Canadian Technical Report of Hydrography and Ocean Sciences

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Regional and headquarters establishments of Ocean Science and Surveys ceased publication of their various report series as of December 1981. A complete listing of these publications and the last number issued under each title are published in the *Canadian Journal of Fisheries and Aquatic Sciences*, Volume 38: Index to Publications 1981. The current series began with Report Number 1 in January 1982.

Rapport technique canadien sur l'hydrographie et les sciences océaniques

Ces rapports contiennent des renseignements scientifiques et techniques qui constituent une contribution aux connaissances actuelles mais que l'on ne trouve pas normalement dans les revues scientifiques. Le sujet est généralement rattaché aux programmes et intérêts du service des Sciences et Levés océaniques (SLO) du ministère des Pêches et des Océans.

Les rapports techniques peuvent être considérés comme des publications à part entière. Le titre exact figure au-dessus du résumé du chaque rapport. Les résumés des rapports seront publiés dans la revue Résumés des sciences aquatiques et halieutiques et les titres figureront dans l'index annuel des publications scientifiques et techniques du Ministère.

Les rapports techniques sont produits à l'échelon régional mais sont numérotés et placés dans l'index à l'échelon national. Les demandes de rapports seront satisfaites par l'établissement auteur dont le nom figure sur la couverture et la page de titre. Les rapports épuisés seront fournis contre rétribution par des agents commerciaux.

Les établissements des Sciences et Levés océaniques dans les régions et à l'administration centrale ont cessé de publier leurs diverses séries de rapports depuis décembre 1981. Vous trouverez dans l'index des publications du volume 38 du *Journal canadien des sciences halieutiques et aquatiques*, la liste de ces publications ainsi que le dernier numéro paru dans chaque catégorie. La nouvelle série a commencé avec la publication du Rapport nº 1 en janvier 1982.

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December 1982

PRE-OPERATIONAL ENVIRONMENTAL MONITORING REPORT FOR THE POINT LEPREAU, N.B., NUCLEAR GENERATING STATION - 1981

by

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ABSTRACT

Smith, J.N., Ellis, E.M., and Bishop, F.J. 1982. Pre-operational monitoring report for the Point Lepreau, N.B., nuclear generating station - 1981. Can. Tech. Rep. Hydrogr. Ocean Sci. 4: vi + 194 p.

The Point Lepreau Environmental Monitoring Program (PLEMP) has been established within the Department of Fisheries and Oceans to assess the environmental impact of radioactive, thermal, and chemical releases from the Point Lepreau, N.B., Nuclear Generating Station located on the Bay of Fundy. During the pre-operational phase of this program, baseline radioactivity levels, in addition to other oceanographic and environmental parameters, were measured in the major environmental reservoirs such as seawater, sediments, marine flora and fauna, soils, terrestrial and aquatic organisms, and atmospheric gases and particulates. These results are presently being used to identify transport pathways for radionuclides through environmental phases in the Point Lepreau region and to determine fluxes of radionuclides along specific pathways. Other studies carried out during the pre-operational phase of this program include a surface and bottom drifter release experiment designed to identify water circulation patterns in the Bay of Fundy, marine and terrestrial ecological baseline surveys, and meteorological experiments characterizing potential dispersion pathways for radionuclides introduced into the atmosphere. Measurements conducted during the operational phase of the reactor lifetime (beginning in late 1982) will be compared to the pre-operational conditions in order to establish the environmental effects of the reactor as a function of time. The ultimate goal of this program is to provide government with a sound scientific basis upon which to assess the environmental implications of the operation of nuclear reactors in coastal regions.

RÉSUMÉ

Smith, J.N., Ellis, E.M., and Bishop, F.J. 1982. Pre-operational monitoring report for the Point Lepreau, N.B., nuclear generating station - 1981. Can. Tech. Rep. Hydrogr. Ocean Sci. 4: vi + 194 p.

Le Programme de surveillance écologique de Pointe-Lepreau (PSEP) a été créé par le ministère des Pêches et Océans afin d'évaluer les répercussions écologiques des rejets radioactifs, thermiques et chimiques de la centrale nucléaire de Pointe-Lepreau située sur la rive de la baie de Fundy. Pendant la phase préopérationnelle actuellement en cours de ce programme, les niveaux de radioactivité de base ont été mesurés dans les principaux réservoirs écologiques que constituent l'eau de mer, les sédiments, la flore et la fauna marines, les sols ainsi que les gazs et les particles de l'atmosphère. Ces mesures sont utilisées avec la détermination d'autres paramètres écologiques et océanographiques pour déterminer les trajets de transport des radionucléides dans l'environment et pour estimer l'importance de certains d'entre eux. Parmi les autres études effectuées pendant la phase préopérationnelle de ce programme, mentionnons une étude à l'aide de bouées dérivantes au fond et en surface visant à caractériser les trajets de transport des radionucléides dans le milieu marin et un relevé du benthos à proximité du point de déversement du réfrigérant qui fournira une base de données permettant de déterminer les répercussions des rejets thermiques sur la diversité et la structure des communautés benthiques. Les mesures effectuées pendant la phase d'exploitation du réacteur (commençant en 1982) seront comparées aux conditions préopérationnelles; on pourra établir les effets écologiques de cette exploitation en fonction du temps. Le but ultime de ce programme est de mettre à la disposition du governement une base scientifique solide permettant d'évaluer les implications écologiques de l'exploitation des réacteurs nucléaires dans les régions côtières.

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1.0 INTRODUCTION

The Point Lepreau Nuclear Generating Station (NGS) located at Point Lepreau, N.B. on the Bay of Fundy (Fig. 1) houses a 660 Mw CANDU reactor. Upon its completion in 1982, the Point Lepreau NGS will become the first nuclear reactor operating on the Canadian coastline. As a result, radioactive, thermal and chemical releases will be principally into the marine environment, thus presenting potential problems which will not have been previously encountered at other nuclear installations in Canada. In response to the unique environmental constraints posed by the proximity of this reactor to the ocean, the Point Lepreau Environmental Monitoring Program (PLEMP) has been established within the Department of Fisheries and Oceans to assess the long term environmental impact of the Point Lepreau NGS.

In contrast to most previous environmental monitoring programs for nuclear reactors which have been primarily concerned with the impact of radioactivity releases on human health, PLEMP has been designed to provide a broader understanding of the distribution of radioactivity throughout the environment. Radioactivity measurements are conducted on samples collected from the major environmental reservoirs such as seawater, sediments, atmospheric gases and particulates and marine flora and fauna. These measurements, in conjunction with the determination of other oceanographic and ecological parameters are being used to identify transport pathways for radionuclides through environmental phases and to determine fluxes of radionuclides along specific pathways.

The monitoring program is divided into pre-operational and operational phases. During the current pre-operational phase of PLEMP, baseline radioactivity and chemical measurements are being performed in order to characterize the background radioactivity conditions within the vicinity of

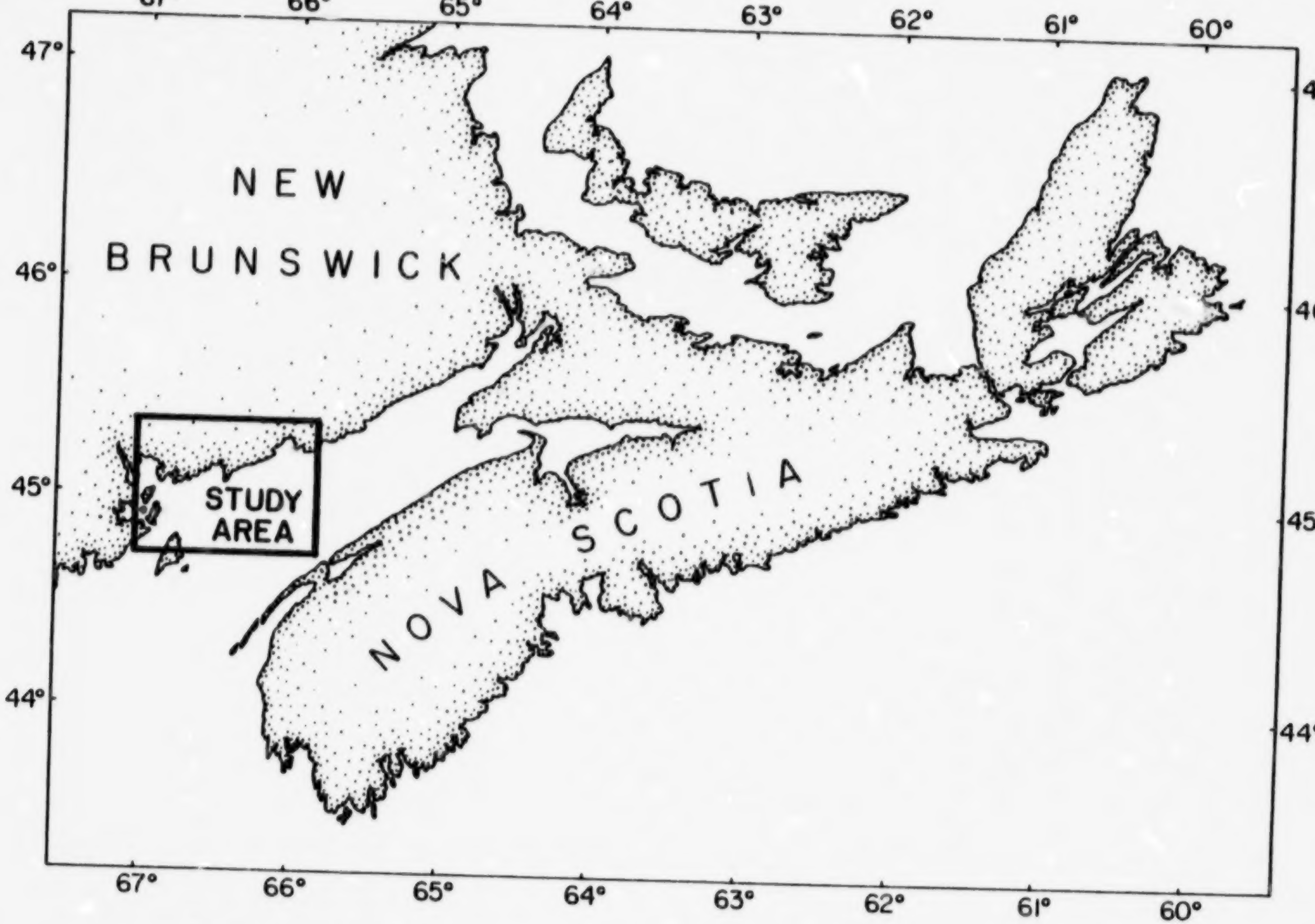


Figure 1. Location of Point Lepreau study area in the Bay of Fundy.

Point Lepreau. Radioactivity measurements are reported in units of Becquerals (Bq). Conversion factors for commonly used radioactivity units are listed in Appendix 1. Measurements conducted during the operational phase of the reactor lifetime will be compared to the pre-operational conditions in order to document any environmental effects associated with the operation of the reactor. The ultimate goal of the program is to provide government with a sound scientific basis upon which to assess the environmental implications of the operation of nuclear reactors in coastal environments.

Responsibility for PLEMP has been assigned to the Atlantic Environmental Radioactivity Unit (AERU), which is a section of the Chemical Oceanography Division located at the Bedford Institute of Oceanography, Dartmouth, Nova Scotia. In addition, a working group on environmental radioactivity, composed of scientists in various disciplines and departments in the Atlantic region (Appendix 2), has been established to ensure that the concerns and responsibilities of different agencies with respect to Point Lepreau are addressed in a coordinated manner by PLEMP. The working group advises AERU on field activities, reviews reports produced during this program and members of the working group act as scientific authorities for contracts administered by AERU. The results of work conducted during 1980 and part of 1981 are described in this report.

2.0 MARINE ENVIRONMENT

Contamination of the marine environment from the Point Lepreau NGS will occur as the result of releases of radioactive fission and activation products both in the plant's liquid effluent and in stack gases. These radionuclides will be dispersed and diluted, spatially redistributed and ultimately accumulate in specific environmental reservoirs. The fate of radionuclides in the environment is governed by a complex interaction of chemical, physical and biological factors. The fact that the components of nearly all ecosystems contain measurable quantities of artifically-produced radioactivity is eloquent testimony to the ubiquitous and persistent nature of man-produced radioactivity. Of primary interest are transport rates of radionuclides between ecosystem components, the mechanisms responsible for this radionuclide movement and the extent to which radionuclides are concentrated within biotic and abiotic components of ecological systems. In order to assess the environmental impact of radioactivity releases it is first necessary to characterize water circulation patterns in the vicinity of Point Lepreau; identify particle transport pathways; establish baseline chemical and radioactivity levels in seawater, suspended matter, sediments and biological phases; and estimate the (pre-operational) fluxes of radionuclides through different environmental phases. Once these objectives have been met, the magnitude of additional radioactivity inputs occuring during the operational phase of the reactor lifetime can be determined and the net accumulation of reactor-derived radioactivity in the major environmental phases can be estimated.

Two additional types of potential contamination of the marine environment from the Point Lepreau NGS are (1) discharges of cooling water at temperature 20°C above ambient levels, and (2) releases of biocides, in the event that these substances are required to reduce fouling in the cooling system. The principal environmental threat posed by these types of contamination is damage to organisms within the vicinity of the cooling water outfall. Comparative assessments of the pre-operational and post-operational structure and composition of benthic communities at Point Lepreau are required to characterize environment stress caused by these releases.

Much of the marine environmental work is conducted during oceanographic cruises in the Bay of Fundy. Five cruises have been carried out by AERU; two in 1979 (Bishop et al., 1980; Smith et al., 1981), one in 1980 (Figure 2) and two in 1981 aboard the CSS Hart (Feb. 23-26; Figure 3) and the CSS Pandora (April 14-21; Figure 4). The results of the 1979 cruises and other studies designed to meet the above objectives have been previously reported (Bishop et al., 1980; Smith et al., 1981) while results of more recent work are reported in the following sections of this report.

2.1 Water Circulation

Circulation in the Bay of Fundy is dominated by a cyclonic (anti-clockwise) gyre with inflow along the coast of Nova Scotia and outflow along the New Brunswick coast passing to the east of Grand Manan Island. In a partially stratified estuary the fresher, less dense water flows out at the surface and ocean water enters at depth. This simple mechanism also operates in the Bay of Fundy where the oceanic sources consist of shelf and slope water which undergoes mixing across the shelf break and enters the Bay of Fundy at depths predominantly below 100 m. About 60% of the freshwater drainage into the Bay of Fundy is derived from the Saint John River. The freshwater plume from the Saint John River extends in a southwesterly

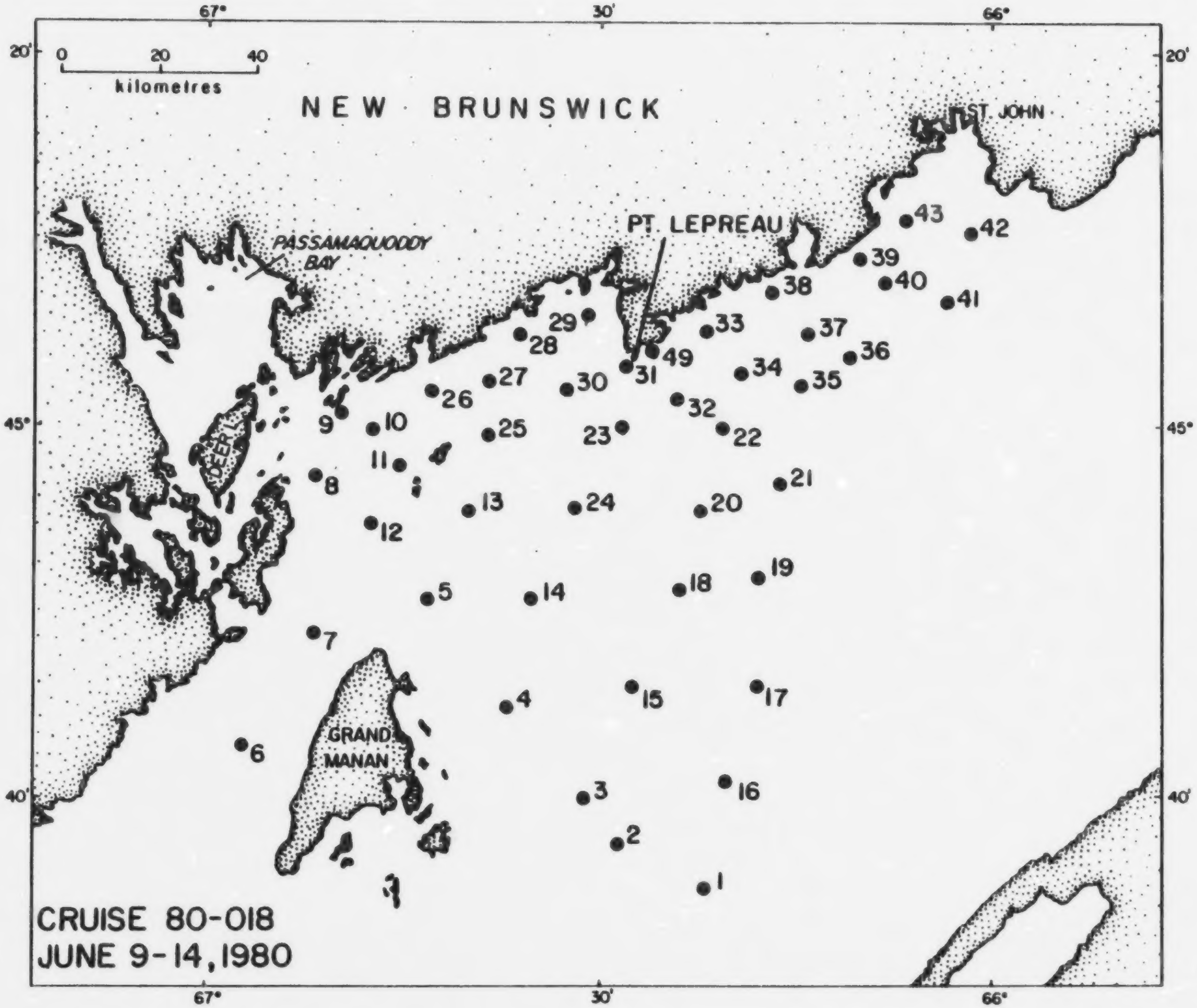


Figure 2. Station locations for cruise 80-018.

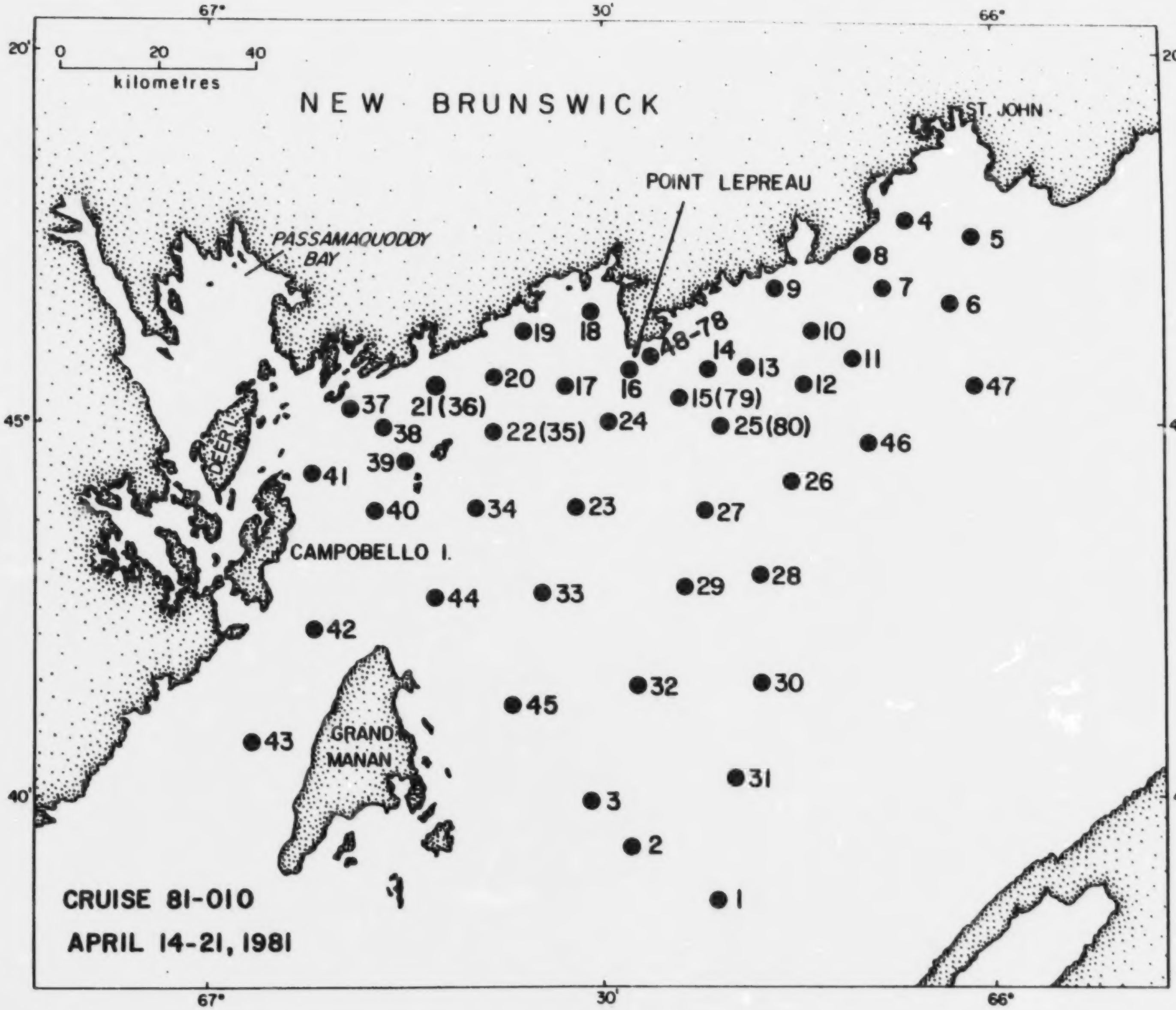


Figure 3. Station locations for cruise 81-500.

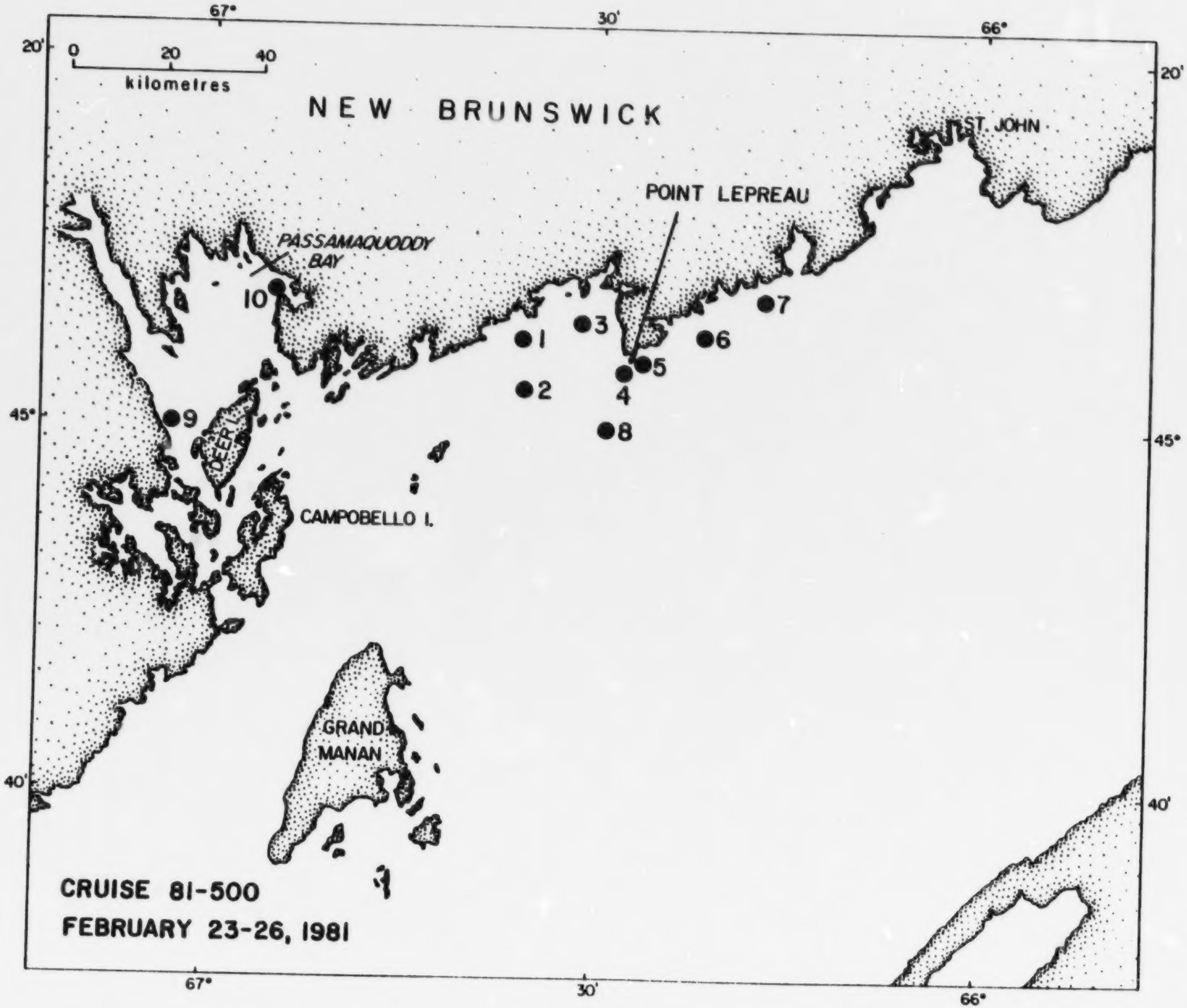


Figure 4. Station locations for cruise 81-010.

direction past Point Lepreau and may reach as far as Grand Manan Island during the peak of the spring runoff. The proximity of the fresh water plume to Point Lepreau may, at certain times of the year result in the release of radionuclides into seawater having reduced salinity. This may affect the chemical speciation of certain radionuclides and their partitioning between dissolved and particulate phases. More efficient mixing and dispersion of radionuclides released from the reactor can be expected during the fall and winter months of low river runoff compared to the spring months when the water column is more stratified. Pollutants injected directly into the freshwater plume, particularly during the spring months, can be expected to travel greater distances, either in solution or in association with suspended matter, prior to dilution by higher salinity bottom water. Some hydrographic data collected during the AERU series of cruises has been previously reported (Smith et al., 1981) and a complete summary will be reported separately (Bugden, in preparation).

2.2 Ocean Drifter Program

Transport of pollutants from the Point Lepreau NGS will be largely controlled by water circulation patterns within the Bay of Fundy. In an effort to characterize these circulation patterns, a drifter program was initiated in October, 1978 involving the continuous release of greater than a thousand surface and bottom drifters during 19781979. The surface drifters are plastic cards designed to float at the water surface with minimal area exposed to the wind and are intended to track possible routes of dissolved or thermal contamination. The bottom drifters are specially shaped, weighted devices designed to float at or near the ocean bottom and

Figure 4. Station locations for cruise 81-010.

are used to track possible routes of sediment-borne pollutants. Both types of drifters are recovered by the general public and returned with details of location, etc., for a $1.00 reward. The purpose of this experiment was to identify transport pathways for radioactivity releases from the Point Lepreau nuclear reactor and to locate sites of pollutant accumulation which will become candidate locations for increased surveillance. The initial results of this program were reported in Bugden (1980) and Smith et al. (1981). During the past year, 21 bottom and 10 surface drifter returns have been reported and these additional results are discussed below.

2.2.1 Surface drifter returns

The additional surface drifter returns raise the overall return percentage from 17.0% to 18.9%. The pattern of additional surface drifter returns (Figure 5) is similar to the previously reported pattern (Figure 6). Four of the additional returns were drifters of a type discovered to be faulty, tending to sink upon release and the data derived from these drifter returns is of dubious significance. The average time adrift for these drifters was 826 days compared to 230 days for the remainder of the additional returns. These four returns comprise the two recovered from the Nova Scotia coast along with two recovered adjacent to the release point. The pattern for the remaining six returns does not significantly alter the conclusions of the initial report.

2.2.2 Bottom drifter returns

The additional 21 bottom drifter returns resulted in a substantial increase in the overall return percentage from 11.5% to 15.3%. Again, the

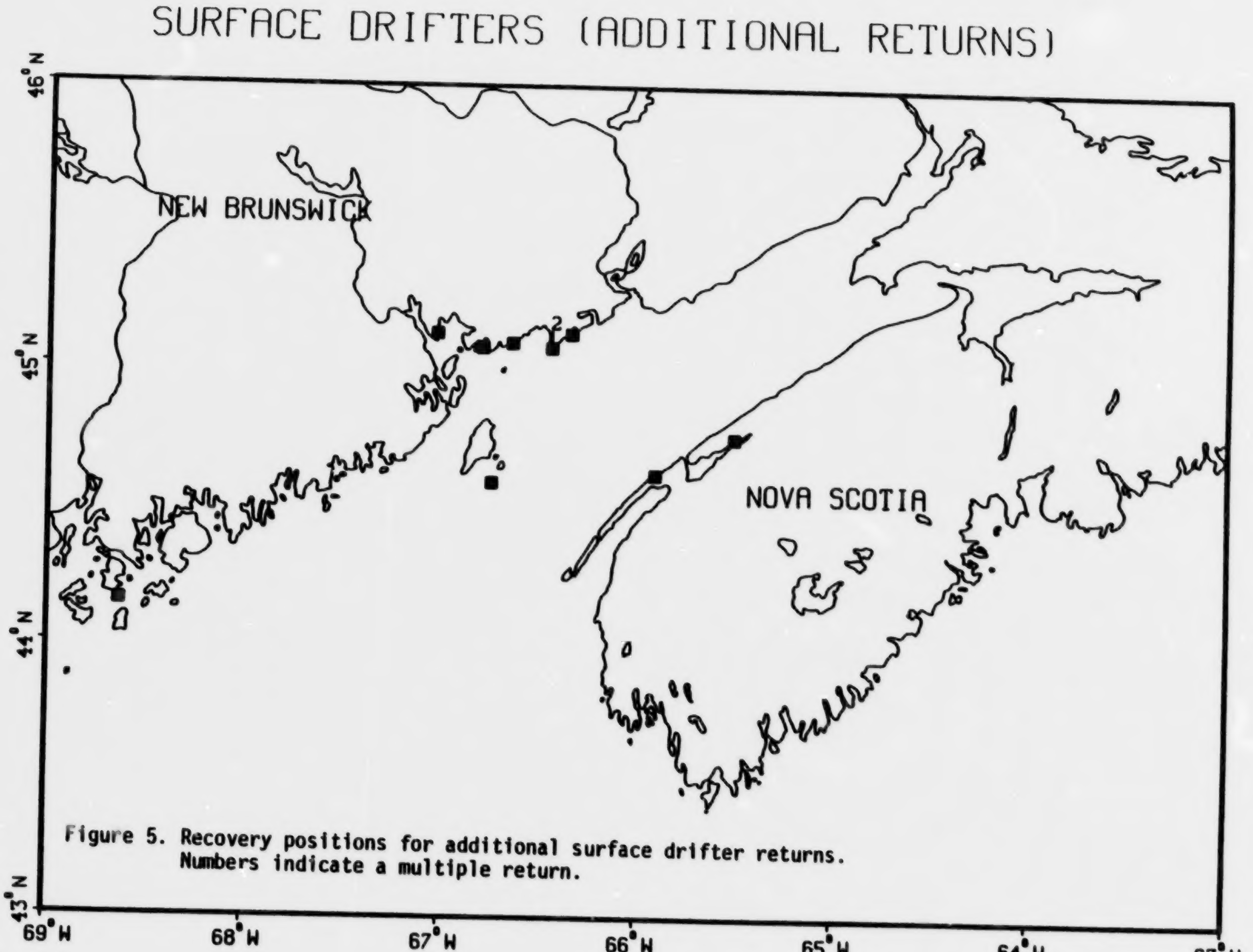
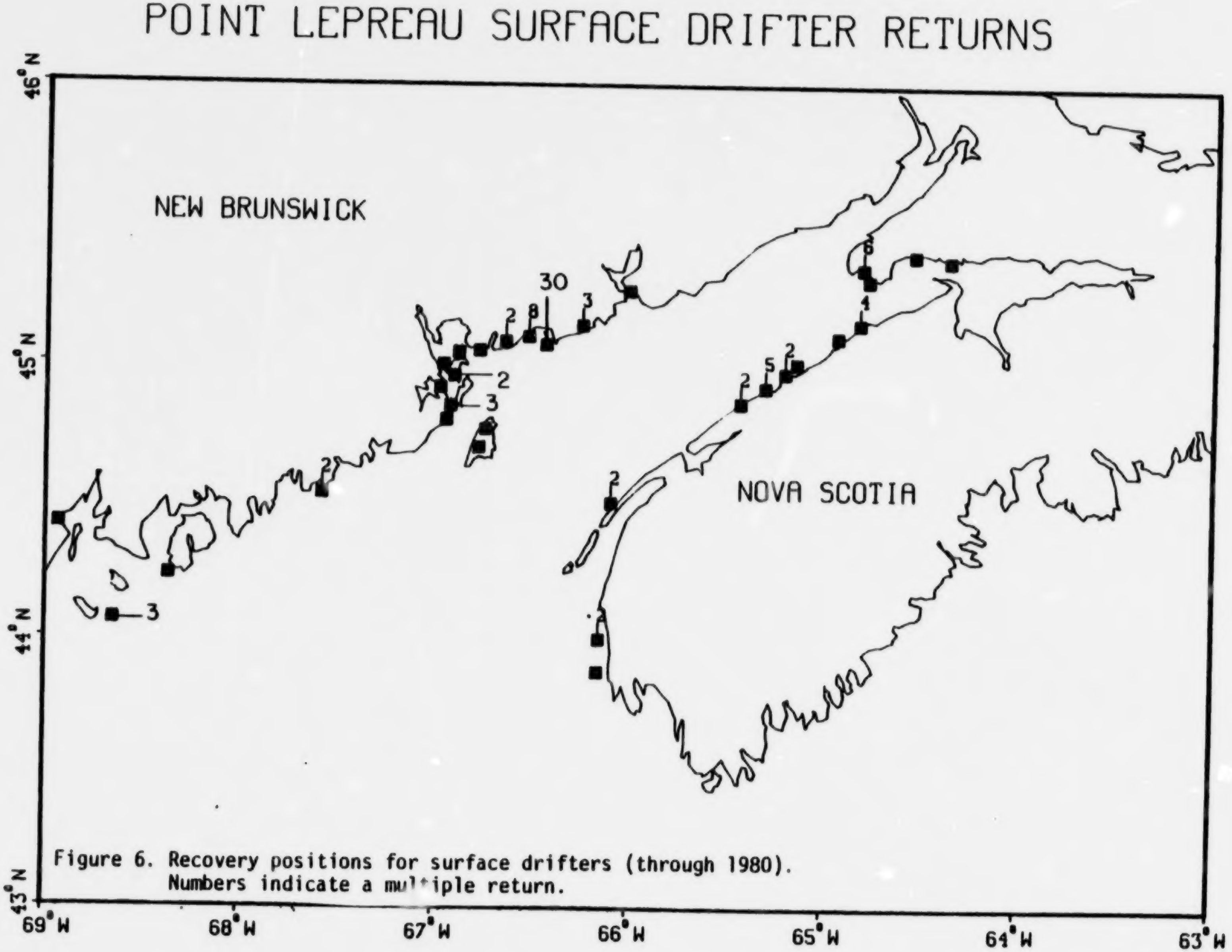


Figure 5. Recovery positions for additional surface drifter returns. Numbers indicate a multiple return.



POINT LEPREAU SURFACE DRIFTER RETURNS

Figure 6. Recovery positions for surface drifters (through 1980).
Numbers indicate a multiple return.

basic pattern of the additional bottom returns (Figure 7) is similar to the previously reported pattern for bottom returns (Figure 8). One of the additional drifters, adrift for 363 days, was recovered on the coast of Maine, much farther south than any previously reported return, but little significance can be attached to a lone recovery. The average time adrift for all additional returns is 491 days, with a minimum drift time of 274 days and a maximum drift time of 1083 days. These additional results raise the total average drift time for bottom drifters from the previously reported value of 167 days to a value of 245 days.

The most significant aspect of the additional bottom drifter data set is its similarity to the previously reported pattern despite the fact that the additional drifters were recovered more than a year after their release. Further, the proximity of the return locations to the release point suggests that the drifters resided in the Maces Bay - Passamaquoddy Bay area prior to their recovery, and indicates that this region is one of pronounced sediment deposition. Seasonal return percentages for all bottom drifters were; 25% in winter (December-February), 39% in spring (March-May), 17% in summer (June-August) and 19% in fall (September-November). Recovery percentages for various drift times for drifters released during different seasons are indicated in Figure 9. The distribution of drift times for drifters released during the spring and winter months exhibits a pronounced maximum at short recovery times while that for drifters released during the summer and fall is more symmetric with respect to the mean drift time. This is an interesting result because it is during the winter and spring that the most severe storms occur in the Bay of Fundy. Hence, it may be concluded that the Maces Bay-Passamaquoddy region is an area of active sediment deposition with the sediment being deposited during the

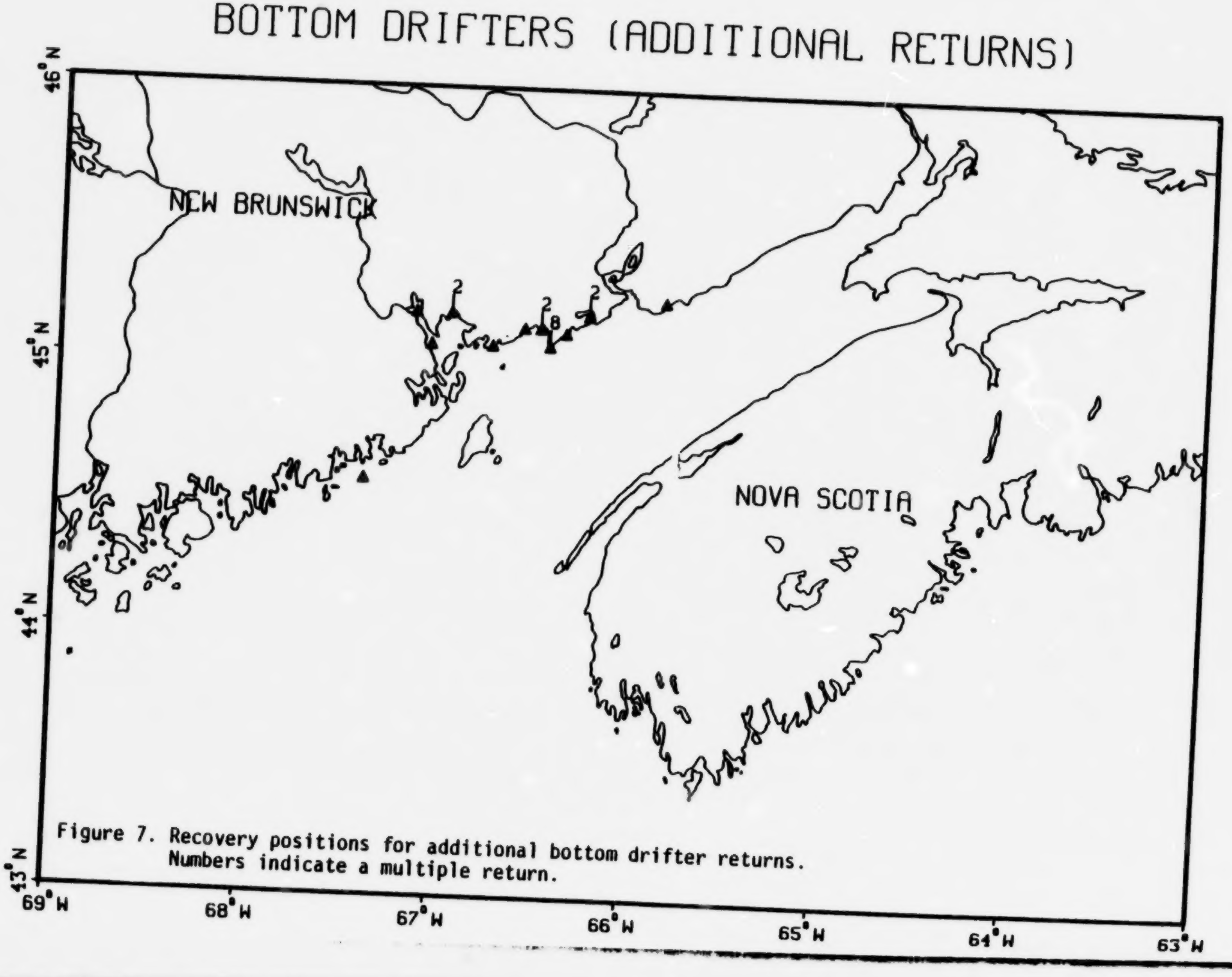


Figure 7. Recovery positions for additional bottom drifter returns.
Numbers indicate a multiple return.

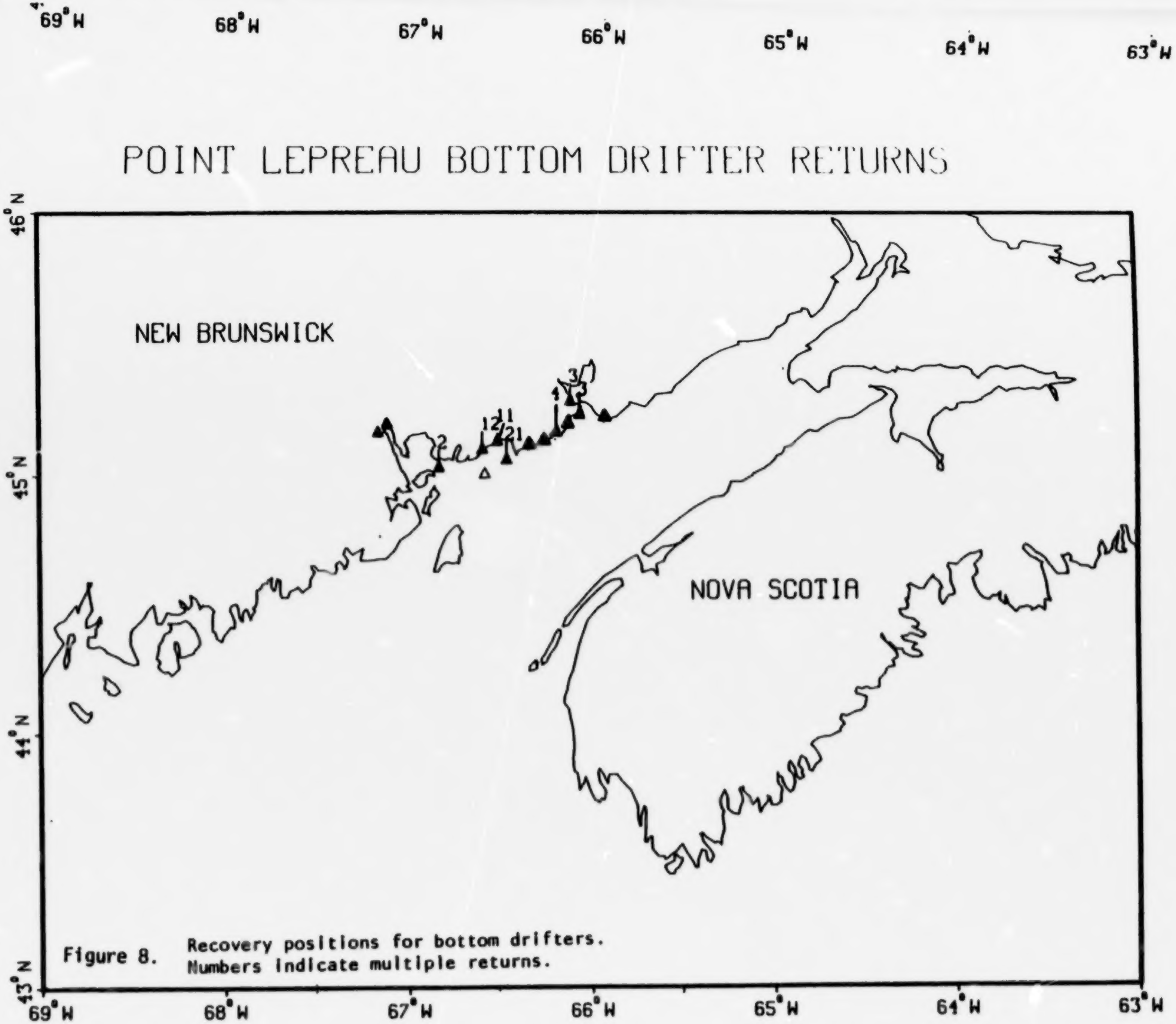


Figure 8. Recovery positions for bottom drifters. Numbers indicate multiple returns.

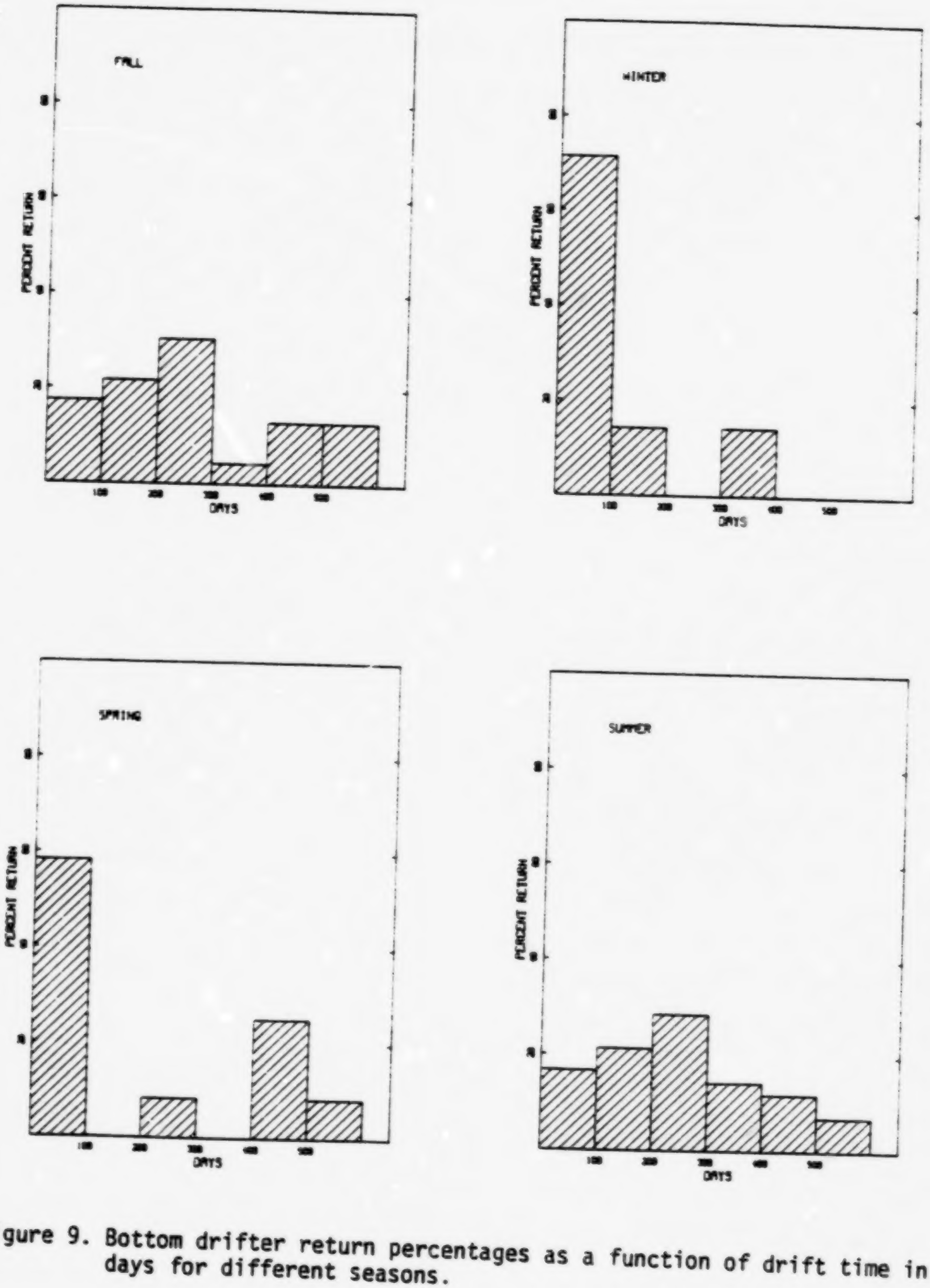


Figure 9. Bottom drifter return percentages as a function of drift time in days for different seasons.

summer and fall and being reworked by winter and spring storms with the accompanying drifters cast ashore and discovered during these periods. Although this pattern of residual bottom drift near Point Lepreau agrees with that found by other investigators, the inferred stronger shoreward transport during the winter and spring months have been previously undetected.

3.0 CHEMICAL OCEANOGRAPHY

The fate of radionuclides released into the marine environment from the Point Lepreau NGS will depend on factors such as their chemical speciation and reactivities and their partitioning between dissolved and particulate phases. The purpose of the chemical oceanographic portion of this program is to develop a better understanding of these factors and of the general water column chemistry in the western Bay of Fundy, through measurements of hydrographic parameters (temperature, salinity), nutrients, trace metals, and suspended particulate levels. Background radioactivity levels are being established in the vicinity of Point Lepreau through measurements of Cs-137, Sr-90, H-3, Pb-210, plutonium and gamma-emitting radionuclides both in the water column and in the sediments. These measurements provide a pre-operational baseline for the assessment of post-operational releases of radioactivity, in addition to radioactive tracer information on water circulation patterns, dispersion rates and particle transport processes in the Point Lepreau Region.

3.1 Radionuclides - Water Column

Seawater sampling procedures and the analytical methods employed in the laboratory for radiochemical separations and detection of environmental levels of radioactivity have been outlined in previous reports in this series. Briefly, large volume (≈50 l) seawater samples are collected using Niskin bottles and the seawater is then passed through (1) a 0.3 micron filter, (2) a Chelex-100 ion exchange column and (3) a KCFC (potassium cobalt ferrocyanide) column. The preweighted filters are subsequently reweighed to determine the SPM (suspended particulate matter) concentration and then analysed in the laboratory for gamma emitting radionuclides.

Cations, including radioactive metals such as Co-60 and Mn-54, are eluted from the Chelex-100 column using aliquots of weak (2N) HNO_3 and HCl and the eluant is subsequently analysed for gamma emitting radionuclides. The KCFC column material is analysed for Cs-137 in the laboratory. Twenty liters of seawater is returned to the laboratory for Sr-90 analysis. Smaller seawater samples are also collected for nutrient, salinity, trace metal and tritium analyses at selected locations. The sampling matrices and nuclear instrumentation used in this program are summarized in Table 1.

3.1.1 Cs-137 and Sr-90 activities in seawater

Cesium-137 ($t_{1/2}$ = 30.2 yr) and Sr-90 ($t_{1/2}$ = 29 yr) are fission products introduced into the environment in significant quantities as a result of nuclear weapons tests conducted in the atmosphere in during the 1950's and early 1960's. The pattern and time scale of Sr-90 deposition has been followed particularly closely because of its long half-life, potentially deleterious biological impact (it follows the same metabolic pathways as calcium) and the existence of relatively direct pathways by which this radionuclide can reach man. Cs-137 has also received increased attention in recent years because of its high radiotoxicity and its usefulness as a tracer for global scale processes. These are the only fission products which will be released from the Point Lepreau NGS which are now generally present at detectable levels in the environment. Pre-operational measurements of these radionuclides in the Bay of Fundy are of interest for two basic reasons; (1) reactor-derived Cs-137 and Sr-90 will generally follow the same environmental transport pathways as their fallout counterparts and detailed studies of the latter species should enable us to identify these pathways and anticipate the magnitude of their net accumulation

Table 1. Summary of nuclear instrumentation used in the AERU laboratory

NUCLIDE	Environmental Phase	Counting Matrix	Detection System
Cs-137	Seawater	KCFC	GeLi and NaI Detectors
Cs-137	SPM	Filter Cartridge (0.3 μm)	GeLi Detector
Cs-137	Sediment	Dried Bulk Sediment	GeLi and NaI Detectors
Sr-90	Seawater, SPM and Sediment	Y-90 on Filter Paper	Alpha/Beta Counter
Gamma Emitters	Seawater	Chelex-100 Eluate	GeLi Detector
Gamma Emitters	SPM	Filter Cartridge (0.3 μm)	GeLi Detector
H-3	Seawater	Gel (Water/Cocktail)	Liquid Scintillation Counter
Pb-120 (Po-210)	SPM, Sediments	Ag,Ni Discs	Alpha Spectrometer (Surface Barrier Detectors)
Pu-239,240	Sediment	Steel Disc	Alpha Spectrometer
Ra-226	Sediment	Ra-222 gas	Gas Phase Scintillation Counter
Ra-226	Sediment	Bi-214 in Dried Bulk Sediment	GeLi Detector

in different environmental reservoirs, (2) the comparatively high background levels of Sr-90 and Cs-137 must be established in order to determine any additional enhancement of these levels caused by releases from the Point Lepreau NGS.

Cs-137 results for seawater samples collected during the June 1980 cruise (#80-018) are listed in Tables 2 and 3. The mean Cs-137 activity in seawater for all stations other than anchor station 49 was 4.24 mBq/l with a range of 3.8 mBq/l to 5.0 mBq/l. This mean Cs-137 activity is close to the mean value of 4.0 mBq/l determined for the May 1979 cruise and greater than the mean value of 2.7 mBq/l determined for the October 1979 cruise. Surface ocean Cs-137 activities have been reported to be in the range of 1 to 20 mBq/l and decrease to background levels with increasing water depth below the surface mixed layer (Bowen et al., 1974, Murray et al., 1978). As has been observed on previous Bay of Fundy cruises, there is no strong correlation between Cs-137 activity and either water depth or geographical location, observations which reflects the efficient tidal mixing in the Bay of Fundy. There was also no significant correlation between Cs-137 activity and either suspended particulate matter concentration or salinity, in contrast to the data set derived from the two 1979 cruises in which a weak ($r = -0.55$, $n = 80$) inverse correlation was found between the Cs-137 activity and salinity. However, the ranges of salinity and Cs-137 activity measured during the 1980 cruise were extremely narrow and the effect of freshwater inputs, possibly characterized by elevated Cs-137 activities in particulate phases, is not resolved in the present data set. Suspended particulate matter concentrations are almost invariably greater in bottom water compared to surface water at each station, probably as a result of

Table 2. Cesium-137 and H-3 results for water samples collected on cruise 80-018.

Sample No.	Station	Water Depth (m)	Salinity (°/..)	SPM (mg/1)	Cs-137 (mBq/1)	H-3 (Bq/1)
80-3400	1	143	32.805	1.20	4.2 ± 0.5	-
80-3401	1	3	32.364	0.42	4.3 ± 0.5	-
80-3406	3	172	32.984	1.41	4.3 ± 0.5	<2.1
80-3407	3	3	31.491	0.88	3.8 ± 0.5	<2.3
80-3412	5	129	32.224	0.76	4.7 ± 0.3	-
80-2313	5	2	31.709	0.60	4.7 ± 0.5	-
80-3418	7	99	32.229	0.61	-	<2.1
80-3420	7	3	31.197	0.50	-	<2.0
80-3421	7	97	32.393	0.94	4.5 ± 0.7	-
80-3423	7	3	31.813	0.62	-	-
80-3424	7	95	32.334	0.90	4.2 ± 0.5	-
80-3425	7	4	31.696	0.75	4.1 ± 0.5	-
80-3426	8	52	32.094	0.56	-	<2.1
80-3428	8	3	31.833	0.37	-	<2.0
80-3430	9	36	31.882	3.53	4.5 ± 0.5	-
80-3431	9	2	31.837	9.15	4.3 ± 0.6	-
80-3438	11	68	32.324	1.18	4.3 ± 0.5	-
80-3439	11	3	31.709	0.89	4.3 ± 0.5	-
80-3440	12	95	32.246	2.38	-	<2.1
80-3441	12	2	31.714	0.48	-	<2.8
80-3445	13	97	32.325	8.19	3.5 ± 0.5	-
80-3446	13	3	31.683	1.09	4.3 ± 0.5	-
80-3449	14	113	32.485	3.38	4.3 ± 0.7	<2.1
80-3450	14	4	31.553	1.58	4.5 ± 0.7	<2.2
80-3460	17	129	32.663	1.34	4.2 ± 0.5	-
80-3461	17	2	31.948	0.92	4.7 ± 0.7	-
80-3464	19	123	32.623	1.21	4.2 ± 0.5	-
80-3465	19	1	32.020	0.66	3.8 ± 0.7	-
80-3472	23	69	32.234	3.07	-	<2.1
80-3473	23	3	31.542	1.33	3.8 ± 0.7	<2.3
80-3474	24	71	32.259	3.96	4.2 ± 0.6	-
80-3475	24	3	31.527	1.33	4.2 ± 0.5	-
80-3486	26	47	32.048	1.94	4.4 ± 0.7	<2.1
80-3487	26	2	31.571	1.84	4.2 ± 0.2	<3.4
80-3498	28	31	31.887	1.66	4.1 ± 0.5	-
80-3499	28	2	31.521	0.87	4.2 ± 0.5	-
80-3504	29	22	31.708	1.81	4.0 ± 0.5	-
80-3505	29	1	31.434	1.63	4.3 ± 0.5	-
80-3509	30	35	32.030	3.65	-	<2.1
80-3510	30	1	31.707	0.98	-	<3.9
80-3515	31	32	31.939	6.32	4.0 ± 0.3	-
80-3516	31	2	31.903	3.68	4.2 ± 0.5	-
80-3519	33	44	31.834	7.42	4.2 ± 0.6	<2.1
80-3520	33	2	31.842	6.12	3.8 ± 0.6	<2.0
80-3523	35	84	32.284	2.79	4.3 ± 0.3	-
80-3524	35	3	31.827	1.22	5.0 ± 0.5	-
80-3530	38	36	31.832	6.18	4.3 ± 0.3	-
80-3531	38	1	31.749	1.15	3.8 ± 0.5	-
80-3532	39	42	31.822	13.92	-	<2.7
80-3533	39	1	31.482	3.35	-	<2.0
80-3536	41	78	32.184	3.21	4.2 ± 0.5	-
80-3537	41	2	31.786	1.62	4.8 ± 0.5	-
80-3541	43	25	31.734	3.40	4.2 ± 0.5	<2.1
80-3542	43	2	31.653	2.26	4.3 ± 0.7	<3.3

Table 3. Surface and bottom Cs-137 activity levels measured in surface and bottom water samples collected at anchor station 49 during cruise 80-018.

Sample No.	Cast Time	Water Depth (m)	Salinity (°/oo)	SPM (mg/l)	Cs-137 (mBq/l)	H-3 (Bq/l)
80-3548	0600	43	31.958	3.55	5.2 ± 0.6	-
80-3549	0600	3	31.853	1.45	4.5 ± 0.6	-
80-3550	0800	39	31.904	8.03	4.9 ± 0.6	-
80-3551	0800	3	31.856	3.83	4.9 ± 0.4	-
80-3552	1000	37	31.884	3.23	4.4 ± 0.6	-
80-3553	1000	3	31.797	2.24	4.5 ± 0.8	-
80-3554	1200	39	31.881	7.38	4.7 ± 0.8	-
80-3555	1200	3	31.818	2.28	5.1 ± 1.0	-
80-3556	1400	42	31.955	2.17	4.4 ± 0.8	-
80-3557	1400	1	31.810	2.14	4.9 ± 0.8	-
80-3558	1600	43	31.940	2.95	5.2 ± 0.8	-
80-3559	1600	1	31.791	1.46	4.9 ± 0.8	-
80-3563	1800	37	31.964	6.56	5.8 ± 0.8	-
80-3564	1800	3	31.886	2.78	5.6 ± 0.8	-
80-3565	2000	36	31.874	6.73	4.7 ± 0.8	-
80-3566	2000	2	31.817	3.23	6.0 ± 0.8	-
80-3570	2200	36	31.852	6.54	3.9 ± 0.8	-
80-3771	2200	3	31.821	3.10	3.9 ± 0.8	-
80-3572	2400	40	31.868	8.27	5.0 ± 0.6	-
80-3573	2400	2	31.787	2.78	-	-
80-3574	0200	41	31.852	6.78	5.3 ± 0.8	-
80-3575	0200	2	31.785	2.50	4.9 ± 0.8	-
80-3576	0400	43	31.871	1.91	4.4 ± 0.6	-
80-3577	0400	2	31.833	2.07	4.4 ± 0.6	-
80-3578	0600	40	31.983	5.50	5.1 ± 0.8	-
80-3579	0600	2	31.946	1.71	4.9 ± 0.4	-
80-3580	0800	38	31.843	13.25	-	-
80-3581	0800	2	31.825	2.09	3.5 ± 0.6	-
80-3582	1000	37	31.832	4.47	3.8 ± 0.8	<2.1
80-3583	1000	3	31.761	2.68	-	<2.4
80-3584	1200	37	31.894	8.88	4.7 ± 0.6	-
80-3585	1200	3	31.824	2.54	3.7 ± 0.6	-

sediment resuspension phenonema. Cs-137 activities on suspended particulate matter are generally below the detection limit of 0.5 mBq/l, which means that on average, less than 10% of the fallout Cs-137 in the water column is associated with SPM. Kupferman et al. (1979) reported that less than 3% of Cs-137 is associated with particles in open ocean waters of the Atlantic Ocean.

Surface and bottom (≈45 m) water samples were collected every two hours over a 30 hour period at anchor station 49, located several hundred meters off the reactor outfall at Point Lepreau. Surface and bottom salinities (Figures 10 and 11) behave similarly as a function of time with surface salinities having values approximately 0.1°/₀₀ lower than the bottom water salinities. Surface water SPM levels are constrained to a comparatively narrow (1 to 4 mg/l) range while bottom water SPM levels are considerably greater, having values as great at 13.4 mg/l. Again, these high bottom water SPM levels are due to sediment resuspension and transport in the nephloid layer. Despite the different characteristics of the surface and bottom waters, the Cs-137 activities are similar in both water masses and exhibit a similar variability with respect to time over the 30 hour sampling period. Clearly, sediment resuspension phenonema do not have a significant effect on the Cs-137 activity in the water column under the conditions prevailing at this location and time. This result is in contrast to data collected at the same location in April 1979 when an out of phase, positive correlation was observed between the Cs-137 activity and the SPM level in bottom water; an observation which was interpreted to reflect possible desorption of Cs-137 from either resuspended sediments or from river-borne material. However, the April 1979 data set was collected during a period of extremely high, spring river discharge when the water

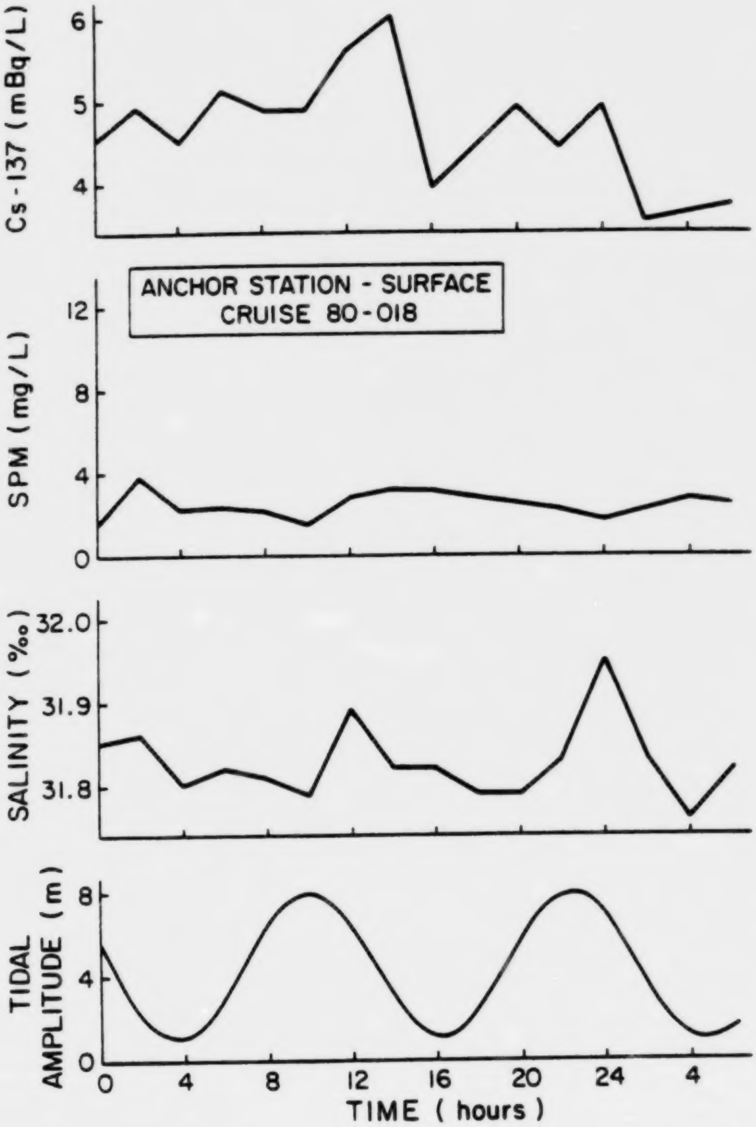


Figure 10. Cesium-137, SPM and salinity plotted as function of time (two hour intervals) for surface water at anchor station 49.

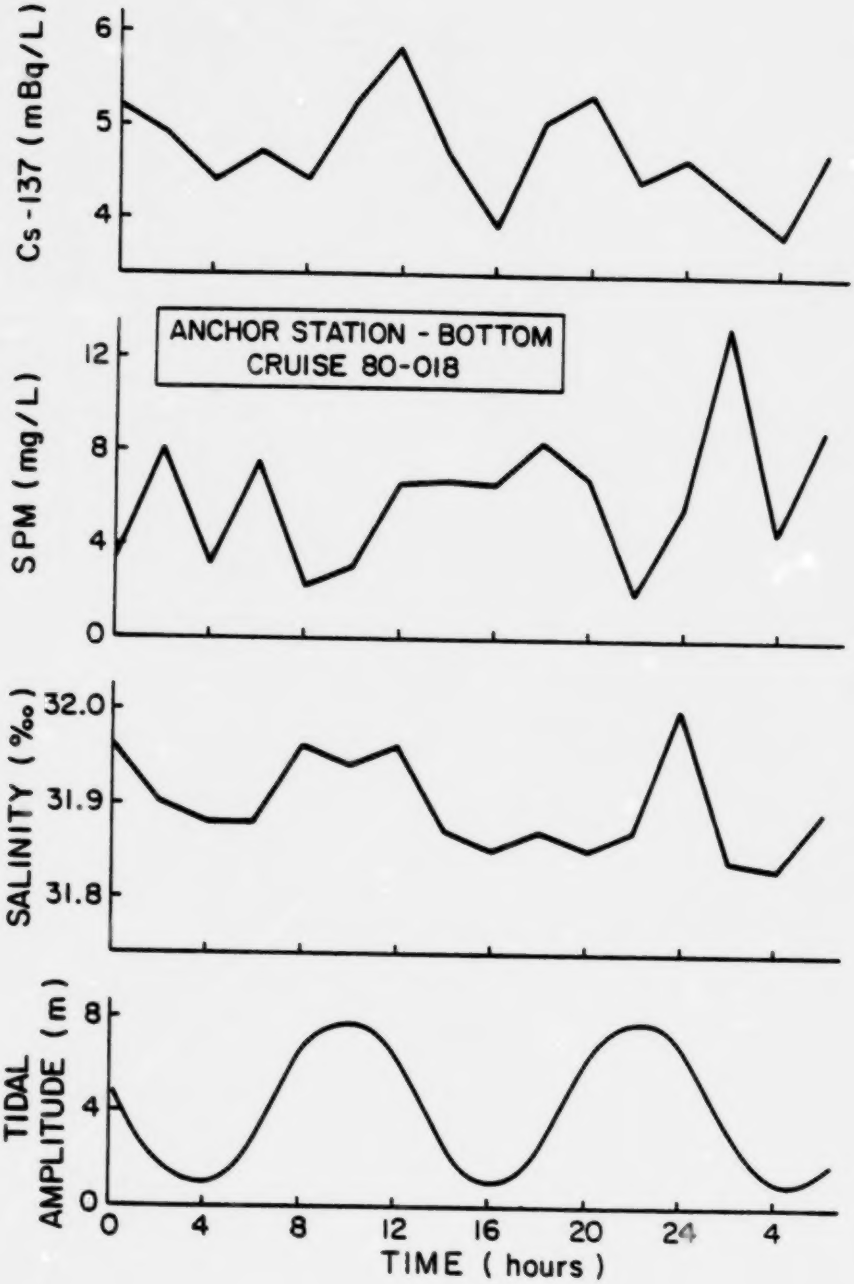


Figure 11. Cesium-137, SPM and salinity plotted as function of time (two hour intervals) for bottom water at anchor station 49.

column was well-stratified. It seems likely that the observed correlation between Cs-137 activity and SPM was due to enhanced inputs of particulate Cs-137 entrained in the Saint John River outflow followed by desorption of the Cs-137 in the vicinity of Point Lepreau. Based on the present set of results, there are vertical gradients in the Cs-137 activity in the water column which are absent under other than high river discharge conditions. It is interesting to note that the temporal variability in the Cs-137 activity at the anchor station over a 30 hour period is as great as the geographical variability exhibited in the Cs-137 activity throughout the western Bay of Fundy. This will become an important consideration in future attempts to detect enhanced levels of Cs-137 due to releases from the nuclear reactor at the cooling water outfall. These results tend to support the contention that tidal mixing will produce rapid dilution of any radioactive plume of radionuclides emanating from the reactor outfall.

The mean Cs-137 activity measured at the ten stations during the February 1981 cruise aboard the CSS Hart was 3.46 mBq/l (Table 4). These data exhibit a weak ($r = 0.45$, $n = 20$) positive correlation with salinity and no significant correlation with SPM concentration. Due to the uncharacteristically warm conditions during this period of the winter, there was a considerable input of fresh water in February leading to greater water column stratification than would be generally expected at this time. However, the Cs-137 activity distribution is similar to that measured during other seasons.

The nutrient results for this cruise (Table 4) lie within the ranges of values measured on previous Bay of Fundy cruises. In particular, silicate concentrations exhibit an inverse relationship ($r = -0.670$, $n = 20$) to salinity (Figure 12) although this correlation is much weaker than

Table 4. Radionuclide and nutrient measurements conducted on samples collected during CSS Hart cruise (#81-500) in 1981.

Sample No.	Station	Water Depth (m)	Salinity (°/oo)	Silicate (g-At/l)	Phosphate (g-At/l)	Nitrate (g-At/l)	Suspended Part. Conc. (mg/l)	Cs-137 (mBq/l)	H-3 (Bq/l)
81-2701	1	35	31.737	8.38	0.81	6.97	33.80	2.5 ± 1.4	<2.1
81-2702	1	1	30.281	7.23	0.58	5.02	1.83	3.4 ± 1.0	<1.9
81-2703	2	37	31.645	5.28	0.68	4.61	1.62	3.8 ± 0.8	<1.9
81-2704	2	1	30.031	6.71	0.53	4.38	1.70	3.6 ± 1.0	<1.9
81-2705	3	23	30.936	8.74	1.34	8.71	11.68	3.2 ± 0.4	<1.9
81-2706	3	1	29.319	11.00	0.96	7.61	2.02	4.1 ± 0.4	5.3 ± 2.3
81-2707	4	29	31.624	8.52	0.89	7.66	5.45	3.6 ± 0.8	<1.9
81-2708	4	1	29.319	10.86	0.81	6.37	10.12	3.3 ± 0.6	2.1 ± 2.3
81-2709	5	36	31.839	9.10	0.92	9.38	9.57	4.4 ± 0.8	<1.9
81-2710	5	1	28.642	11.83	0.93	10.22	2.31	4.0 ± 1.0	4.8 ± 2.3
81-2711	6	42	31.754	7.61	0.77	7.78	33.50	3.4 ± 0.4	<1.9
81-2712	6	1	26.694	10.90	0.68	8.82	10.60	3.4 ± 0.6	<1.9
81-2713	7	35	31.696	9.35	0.90	8.62	3.68	3.9 ± 0.6	<3.9
81-2714	7	1	24.792	12.63	0.75	8.45	2.76	2.2 ± 0.6	2.5 ± 2.1
81-2715	8	61	32.045	7.66	0.77	7.68	3.05	3.6 ± 0.8	<2.1
81-2716	8	1	29.863	9.84	0.78	7.94	0.55	3.0 ± 0.6	3.7 ± 2.1
81-2717	9	40	31.025	9.72	0.85	9.16	1.95	3.5 ± 0.8	<2.6
81-2718	9	1	30.164	10.89	0.84	9.61	5.99	3.7 ± 0.6	5.1 ± 2.1
81-2719	10	8	29.820	8.91	0.70	6.64	1.20	3.7 ± 1.0	<3.7
81-2720	10	1	28.153	-	-	-	2.08	2.8 ± 0.6	3.7 ± 2.1

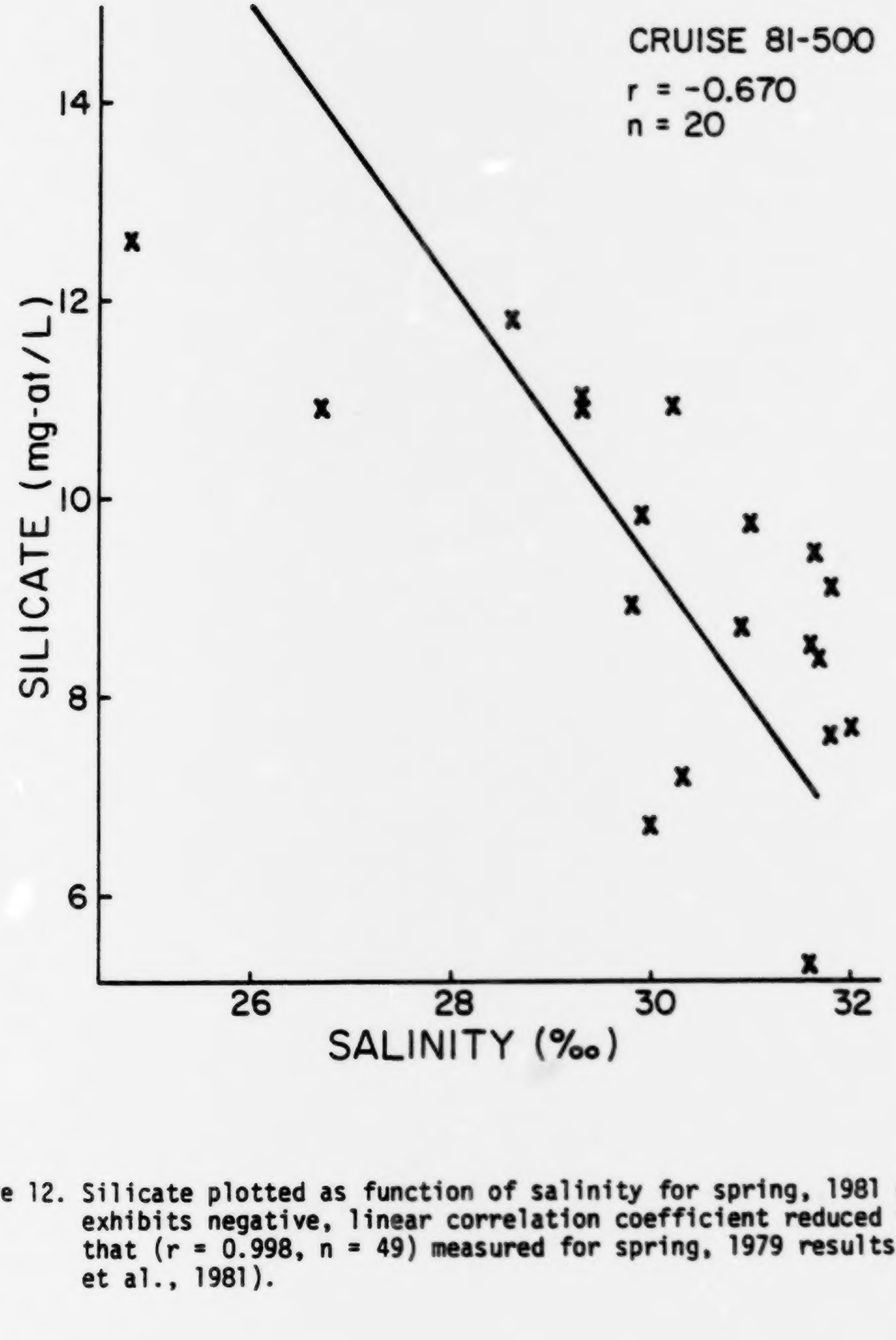


Figure 12. Silicate plotted as function of salinity for spring, 1981 cruise exhibits negative, linear correlation coefficient reduced from that (r = 0.998, n = 49) measured for spring, 1979 results (Smith et al., 1981).

that (r = -0.998, n = 49) measured for samples collected during the May 1979 cruise. These correlations result from the mixing of freshwater, characterized by elevated silicate concentrations, with seawater having reduced silicate concentrations.

3.1.2 Gamma-emitting radionuclides

With the exception of I-131 and Cs-137, the dissolved gamma-emitting radionuclides listed in Table 11 are all extracted from seawater by adsorption onto Chelex-100 cation exchange resin during shipboard sampling operations. The retention efficiency of these radionuclides, determined by following the passage of NBS mixed gamma radionuclide standards through a Chelex-100 column, averages 90%. The radionuclide are eluted from the column by successive additions of 2N HCl and HNO_3. The detection limits listed in Table 5 for each radionuclide are based on an assumed water volume of 36 l with the eluted samples counted for 1500 m, one month after the initial sample collection. For the 20 samples that have been analysed, no radionuclides were present above the detection limit.

Gamma emitting radionuclides are measured on SPM by direct counting of filter cartridges (0.3 μm) on the Ge(Li) detector. Standards were prepared by direct spiking of the filters with a mixed gamma radionuclide standard. The detection limits for each radionuclide listed in Table 5 have been calculated assuming an SPM concentration typical of Bay of Fundy water of 4.7 mg/l, a seawater volume of 36 l, and a counting time of 1500 m, one month after sample collection. Only Cs-137 was detected in several of the 23 particulate samples from the Bay of Fundy. Although many gamma-emitting radionuclides were produced during nuclear weapons tests

Table 5. Detection limits for gamma emitting radionuclides in dissolved and particulate phases. Calculation based on seawater volume of 50 l, SPM concentration of 2.1 mg/l, and counting time of 1500 min one month after sample collection.

Nuclide	Half-life (Days, Years)	Detection Limit Suspended Matter (mBq/l)	Dissolved (mBl/l)
Ag-110 m	250.4d	0.60	5.1
Ce-141	33.0d	1.70	11.6
Ce-144	284.4d	0.51	31.6
Co-58	71.2d	0.70	9.6
Co-60	5.27y	0.60	4.3
Cr-51	27.7d	10.00	64.5
Cs-134	2.06y	0.56	1.0
Cs-137	30.0y	0.60	0.5
I-131	8.06d	14.00	-
Mn-54	321.6d	0.60	3.7
Nb-95	35.2d	1.00	6.8
Ru-103	39.4d	1.00	6.6
Ru-106	368.0d	6.30	38.5
Zn-65	244.1d	1.20	29.6
Zr-95	65.0d	1.30	19.9

conducted in the 1950's and 1960's, their relatively short half-lives have resulted in their radioactive decay to undetectable levels by 1980.

3.1.3 Tritium results

Tritium is produced in heavy water reactors by neutron activation of deuterium and is released in far greater quantities in stack gases and liquid effluent than any other radionuclide. Tritium readily exchanges with water and consequently follows the water cycle both in organisms and within the environment. Owing to the low energy of the beta particle emitted by tritium and its rapid dilution and elimination from organisms this radionuclide is considered to be one of the least hazardous radionuclides and its maximum permissable concentration in drinking water is greater than for any other radionuclide. Nevertheless, the release rate of tritium from the Point Lepreau NGS (≈ 5000 curies/year) will be orders of magnitude greater than that for any other radionuclide and the monitoring of the tritium "plume" in the water column is an important component of the environmental surveillance program.

Prior to the early 1950's and the onset of large scale injection of tritium into the atmosphere from the testing of nuclear weapons, the principal source of terrestrial and oceanic tritium was the stratosphere where it is produced by the action of high energy cosmic rays in spallation reactions with nitrogen. Although Libby (1946) first proposed the existence of these reactions, it was not until 1951 that tritium was detected in Norwegian waters (Grosse et al., 1951). A hundred fold increase in tritium in levels in the environment occurred as a result of nuclear weapons tests conducted in 1962 and 1963, nearly all of which is in the form of tritiated water. Having a half life of 12.3 years, the tritium transient in the surface and "younger" waters of the oceans provides a powerful probe for dynamic studies of circulation processes over time periods of 50 years. Tritium concentrations in north Atlantic surface water peaked in 1964-65 and have since decreased by a factor of more than five (Dreisigacker and Roether, 1978). Measurements of the distribution of tritium along the Atlantic coast made between 1963 and 1967 indicated that the tritium activity range in coastal waters was between 3.4 and 47.8 Bq 1^{-1} (Leventhal and Libby, 1970). Increased tritium activities at more inland stations are due both to smaller dilution factors by oceanic water vapour and to a higher activity of re-evaporated continental moisture. North Atlantic surface water samples collected in the early 1970's off Newfoundland exhibited tritium levels of the order of 1 Bq/l (Ostlund et al., 1974).

Water samples collected for tritium analysis in the Bay of Fundy during cruises 80-018 and 81-500 were distilled under vacuum and counted for 500 minutes using a liquid scintillation counter and methods outlined in previous reports. Tritium free water, obtained courtesy of Dr. H. Gote Ostlund (University of Miami) or Baffin Bay deep water (<0.2 Bq/l), obtained courtesy of Dr. E.P. Jones (Bedford Institute of Oceanography), was distilled in the same manner and counted during each run. Counting efficiency standardization was performed with every sample set with a dilution of the National Bureau of Standards tritium standard reference material 4926-C. Counting efficiencies were in ranged from 25.5% to 26.1% with blank count rates varying between 5.8 and 6.1 counts per minute. The detection limit (L_d) at the 95% confidence level, ranges from 3.5 to 4.1 Bq/l depending on blank count rates and counting efficiency. The results for cruises 80-018 and 81-500 are given in Tables 2, 3 and 4 with all samples having tritium levels below or just marginally above the detection limits.

The mean tritium activity for samples collected on cruise 79-007 was 2.7 ± 3.0 Bq/l and the mean tritium activity for samples collected on cruise 79-027 was 1.7 ± 2.7 Bq/l. The present set of results is not significantly different from the previously performed measurements. No unusual seasonal transients or geographical anomalies have been noted in tritium concentrations within the Bay of Fundy although present day environmental levels of tritium in the study area are below or bordering on the detection limit for the analytical technique employed by AERU.

3.2 Sediment studies

Radionuclides released from the Point Lepreau NGS will be partitioned between soluble and suspended particulate matter phases to an extent depending on their respective particle-water distribution coefficients. The more particle-reactive radionuclides such as Co-60 and Zn-65 will follow the transport pathways for particulate matter through the water column and will ultimately be deposited in the sediments. Sites of net sediment deposition within the western Bay of Fundy warrant investigation because they may ultimately record the historical record of particle-reactive radionuclide releases from the Point Lepreau NGS. Further, knowledge of radionuclide sediment inventories is critical to the determination of the availability of these radionuclides for uptake by bottom dwelling organisms and to assess the importance of post-depositional remobilization of radionuclides by particle resuspension, bioturbation and diffusion. Finally, the sediment inventory of radionuclides may represent a significant fraction of the total budget for radionuclide releases from the Point

Lepreau NGS which must be determined in order to estimate net radionuclide fluxes through different environmental phases.

3.2.1 Analytical methodology

Classification of the sediments on the basis of their relative sand and mud content gives the distribution pattern for the western Bay of Fundy indicated in Figure 13 (Loring, 1979). The finest grain sediments are found in the Quoddy region, west of Point Lepreau and extending in a lobe south to Grand Manan Island which is generally consistent with the distribution pattern of LaHave clays (Fadar et al., 1977). Particle size is the main factor controlling trace metal concentrations in the Bay of Fundy because the adsorption of metals onto the detrital host mineral particles increases with the particle surface area. Therefore the finer offshore sediments in the Quoddy regions are a sink and an enrichment areafor trace metals introduced from both natural and industrial sources such as the St. Croix drainage basin, coastal industries and offshore dredge disposal sites (Loring, 1982).

Sediment samples were collected at all stations during the AERU Bay of Fundy cruises using both Van Veen and Shipek grab samplers in order to identify fine-grained depositional regimes. Box cores and Leheigh gravity cores were subsequently collected at stations selected on the basis of the grab samples. The box cores were sampled at 1 cm intervals on board the research vessel (CSS Dawson) and the gravity cores were stored upright for transport to the laboratory where they were subsequently extruded and sub-sampled at 1 cm intervals. The water content of the samples was routinely measured as a function of sediment depth in order to determine the integrated accumulation of sediment solids in each core as a function of

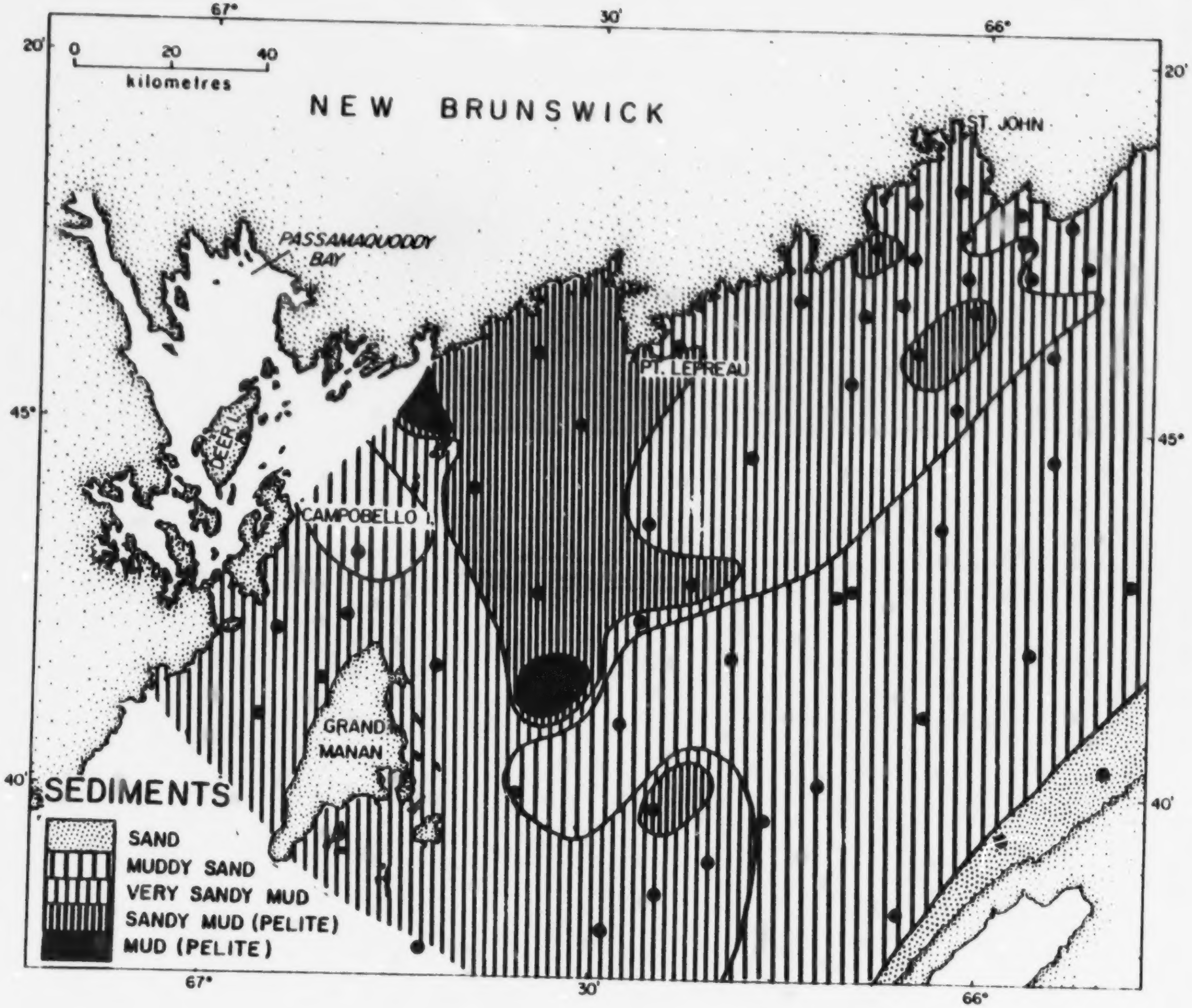


Figure 13. Particle size distribution of sediments in the western Bay of Fundy reproduced from Loring (1979).

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sediment depth. Pb-210 analyses were performed by deposition of its granddaughter Po-210 and tracer Po-208 onto silver or nickel discs from acid-digested sediment samples and Po-210 and Pb-210 activities were determined by alpha spectrometry following the methods outlined in Smith and Walton (1980). Radium-226 activities were determined by gamma spectrometry. Cs-137 activities were determined by non-destructive analysis of dried sediment samples using Ge(Li) and hyper-pure Ge detectors interfaced with a multi-channel analyser.

Plutonium is a fallout radionuclide whose distribution in the environment provides information regarding the transport of particle-reactive radionuclides from the Point Lepreau NGS. Analytical procedures which have been recently developed in the AERU laboratory begin with the addition of a calibrated tracer (Pu-242 or Pu-236) to a 10 g sample of sediment. The plutonium is extracted into a mixture of nitric and hydrochloric acids and passed through an anion-exchange column (Bio-Rad AG1X4, 100-200 mesh). Several purification steps are followed to remove interfering radionuclides and Pu-238 and Pu-239, 240 isotopes are electrodeposited onto stainless steel discs and measured by alpha spectrometry. The activity level of Pu-241 can also be measured through detection of its daughter product, Am-241, after an ingrowth period of 1 to 2 years. For the current background levels of 0.0001 cpm in the region of interest in the alpha particle, surface barrier detectors, and for a chemical yield of 70%, a counting efficiency of 30% and a counting time of 5000 minutes, the present detection limit for Pu-239,240 is 9 mBq/kg of sediment (95% confidence limit).

reproduced from Loring (1979).

3.2.2 Pb-210 and Cs-137 sediment-depth profiles

Pb-210 ($t_{1/2}$ = 22.3 yr) is a naturally-occurring member of the U-238 decay series which is introduced into coastal waters at a relatively constant rate as a result of the radioactive decay of Rn-222 in the atmosphere. Pb-210 is rapidly scavenged by particulate matter in the water column and accumulates as an unsupported excess in the sediments. Measurements of the excess Pb-210 activity, $A_{Pb\text{-}210_{ex}}$ (equal to the total Pb-210 activity minus the supported Pb-210 activity due to radioactive decay of Ra-226 within the sediments), as a function of sediment depth can be used to determine sedimentation rates, rates of sediment mixing by bioturbation, or to identify anomalous depositional events associated with sediment slumping (Smith and Walton, 1980). In contrast, Cs-137 ($t_{1/2}$ = 30 yr) is a fission product initially introduced into the environment during the 1950's as a result of nuclear weapons tests conducted in the atmosphere. Inputs of Cs-137 to the water column were maximized during the early 1960's, the period of most intensive weapons testing, and the timing of these weapons tests may be recorded as peaks in the Cs-137 activity profile in undisturbed sediments. Cs-137 measurements in sediment cores complement Pb-210 studies of sediment transport processes and, in addition, provide baseline data which will be used to assess Cs-137 inputs from the Point Lepreau NGS.

The excess Pb-210 inventory, $I_{Pb\text{-}210_{ex}}$ (integrated quantity of excess Pb-210 in a core) is proportional to the total flux of excess Pb-210 to the sediments, which, in turn, is a function of the sedimentation rate of fine-grained particulate material. The excess Pb-210 inventories in the western Bay of Fundy can be regionally characterized by low (<30 dpm cm^{-2}), medium (30 dpm cm^{-2} - 100 dpm cm^{-2}) and high (>100 dpm cm^{-2}) excess

Pb-210 inventories as illustrated in Figure 14. The highest Pb-210 inventories are found in the Quoddy region, near the northern approaches to Passamaquoddy Bay, where the sediments consist of fine grained muds or pelites (Figure 13). The fine-grained sediments in this region constitute a net depositional sink, both for Pb-210 and other particle reactive radionuclides, in addition to particle-reactive trace metals such as Zn and Hg, as indicated in Loring (1982). In contrast, the sediments in the immediate vicinity of Point Lepreau which grade into sandy muds, are characterized by lower inventories of Pb-210. This latter result is consistent with the inhibition of deposition of fine-grained material as the result of the strong residual bottom currents in this region.

Sediment profiles for Pb-210 exhibit considerable variability in the western Bay of Fundy. The profiles in Figure 15 are from cores 7145, 7146, 3560, 3567, all of which were collected at Station 49, approximately 1 km from the cooling water outlet for the Point Lepreau NGS. The supported Pb-210 (dotted line, Fig. 15) is due to the in situ radioactive decay of Ra-226 within the sediments and must be subtracted from the total Pb-210 activity to determine the excess Pb-210 activity. Cores 7145 and 7146 were collected in comparatively coarse-grained sediments and the effect of particle size is reflected in the very low excess Pb-210 activities in each core. Cores 3560 and 3567 were collected in finer-grained sediments several hundred meters away from the coarser grained sediment regime and these cores contained higher inventories of excess Pb-210. The deeper penetration of excess Pb-210 in cores 3560 and 3567, to depths greater than 20 cm, is a result of downward mixing of surface sediments caused by the feeding, burrowing and irrigation activities (bioturbation) of bottom dwelling organisms. Deposition of reactor-derived radionuclides

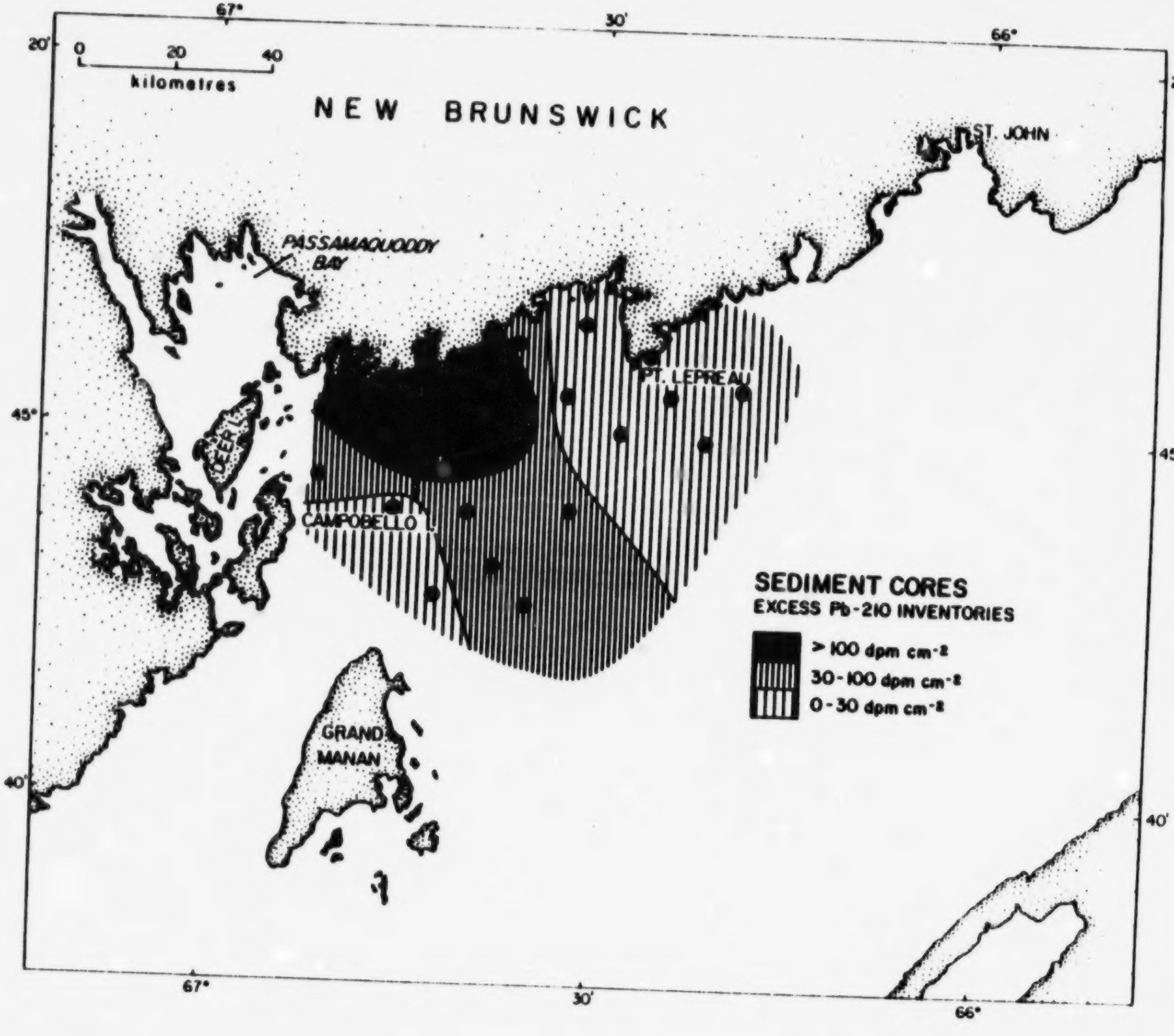


Figure 14. Excess Pb-210 inventories in sediment cores (filled circles) collected

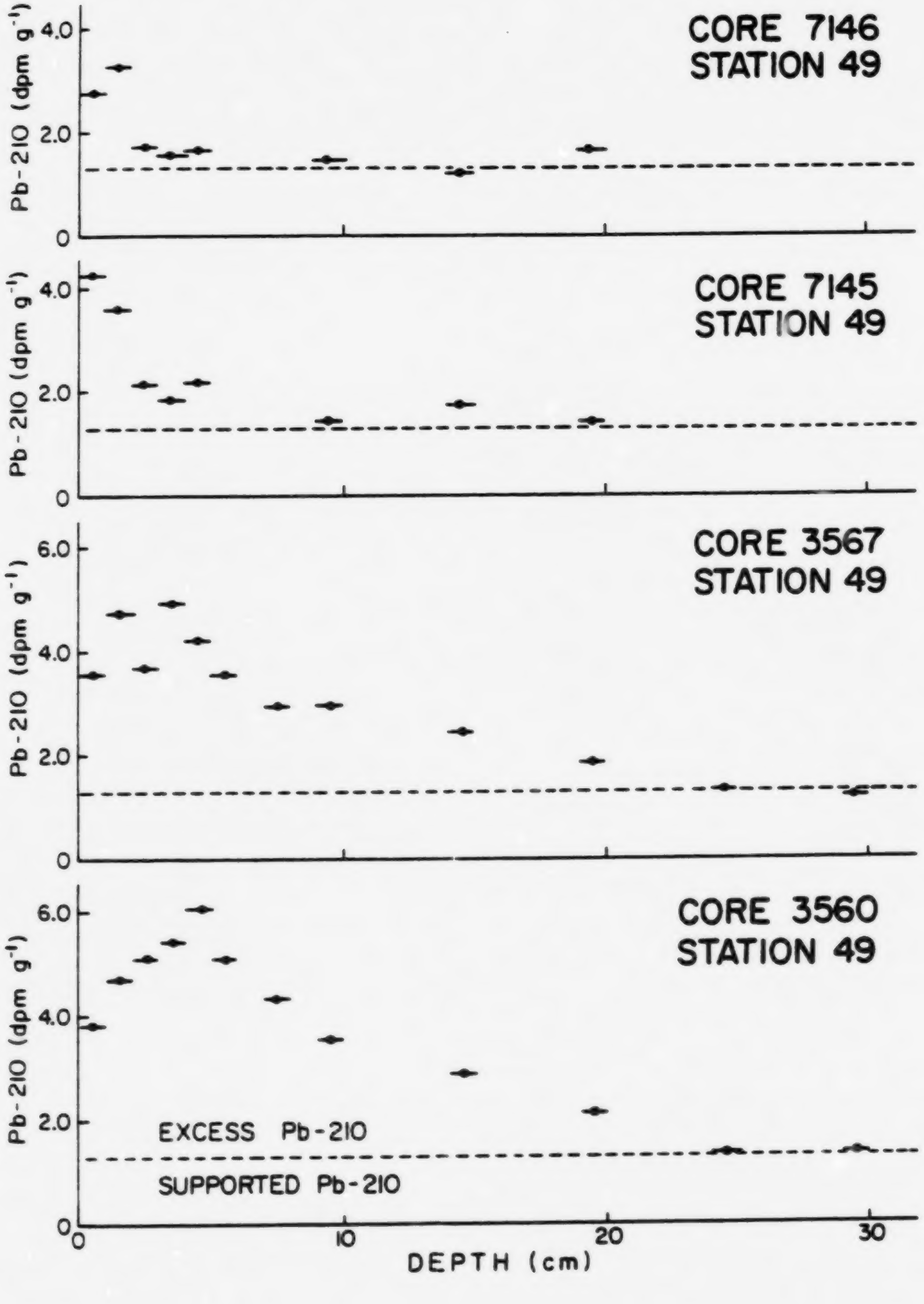


Figure 15. Lead-210 versus sediment-depth profiles for cores collected near the reactor cooling water outfall. Excess Pb-210 is difference between total Pb-210 and supported Pb-210 (dotted lines) determined from Ra-226 measurements.

would be enhanced in this latter sedimentary regime, but as a result of bioturbation these radionuclides would probably be rapidly transported to depth in the sediment column. The sub-surface Pb-210 maxima in cores 3560 and 3567 may be the result of the dynamic interaction between bioturbation and sediment resuspension processes.

For the case in which the sedimentation rate is constant as a function of time and there is negligible post-depositional redistribution of sediment as the result of bioturbation, slumping or resuspension, then the excess Pb-210 activity will decrease exponentially as a function of sediment depth. Measurements of the slope of the excess Pb-210 profile can be used to estimate the sedimentation rate, as outlined in Smith and Walton (1980). Bioturbation or other forms of sediment mixing will produce downward transport of Pb-210 within the sediment column, resulting in a decrease in the slope of the Pb-210 profile and an increase in the calculated sedimentation rate. The sedimentation rate estimated in this case represents an upper limit on the actual sedimentation rate which may be considerably lower, and in some cases, negligible (Benninger et al., 1979; Santchi et al., 1980).

The trend in "apparent" sedimentation rates, estimated from Pb-210 profiles in cores collected on transects running westerly from Point Lepreau, to the northern entrance to Passamaquoddy Bay (Figure 16) is illustrated in Figures 17 and 18. A gradual increase in the apparent sedimentation rate is evident between station 34 and station 30. This sedimentation rate increase from 0.11 cm/yr (Sta. 34) to 0.18 cm/yr (Sta. 30) is consistent with an increase in the clay-silt (mud) components of the sediments. However, the Cs-137 data indicate that some sediment mixing, probably as the result of bioturbation, has occurred in these sediments. This

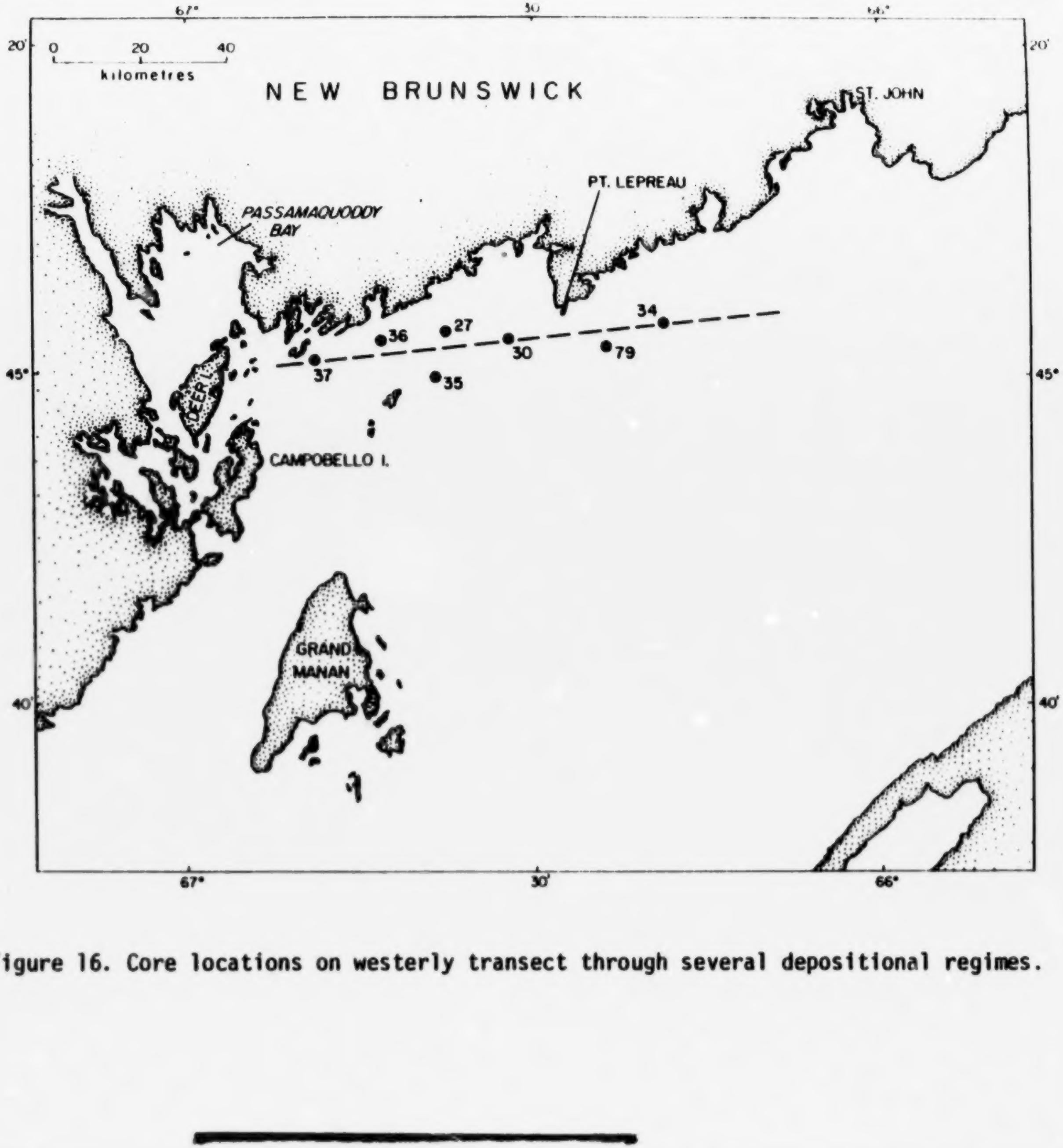


Figure 16. Core locations on westerly transect through several depositional regimes.

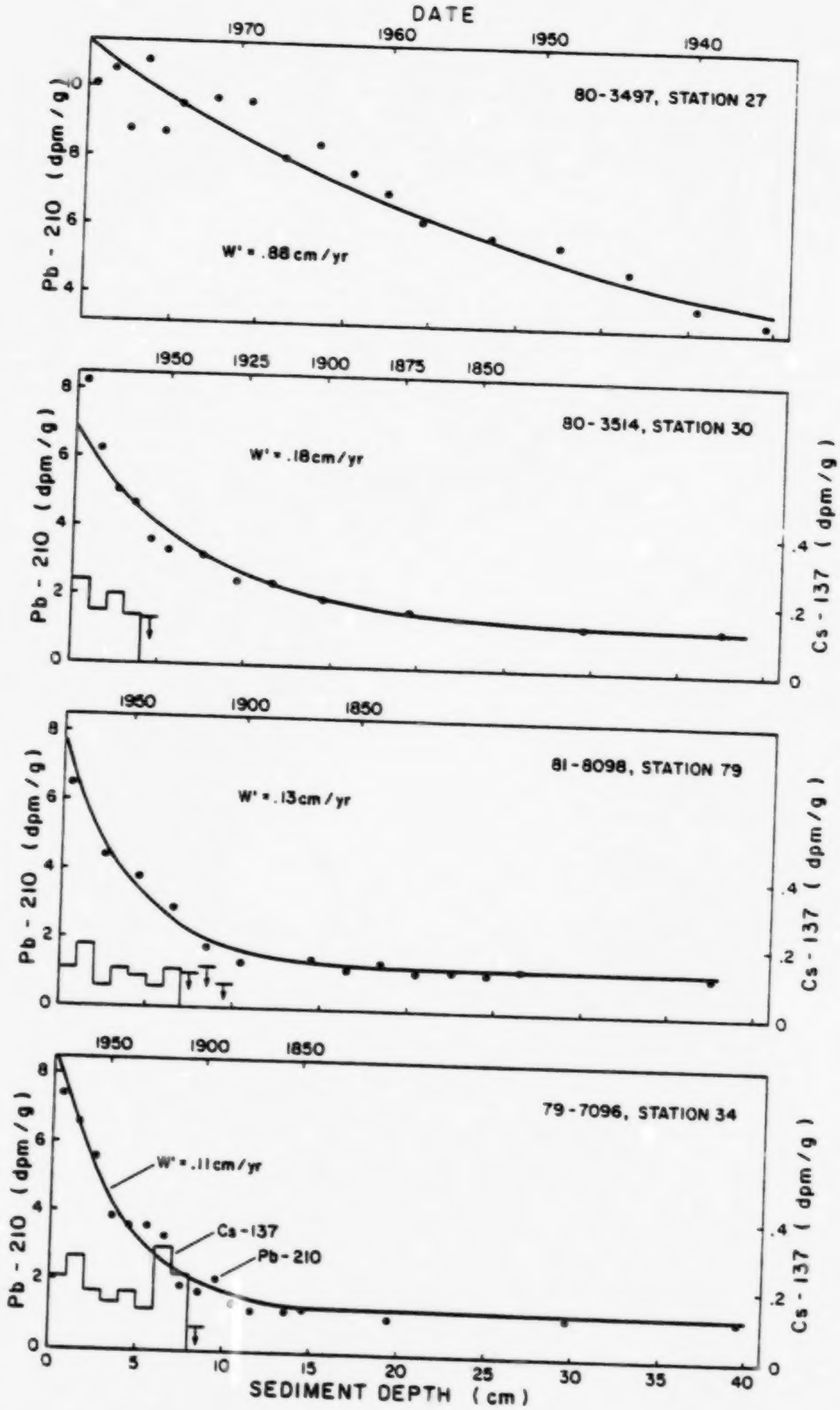


Figure 17. Pb-210 (circles) and Cs-137 (lower solid lines) profiles for cores collected on transect given in Figure 16. Apparent sedimentation rates (W') estimated from least squares, exponential fit (solid line) to the Pb-210 data.

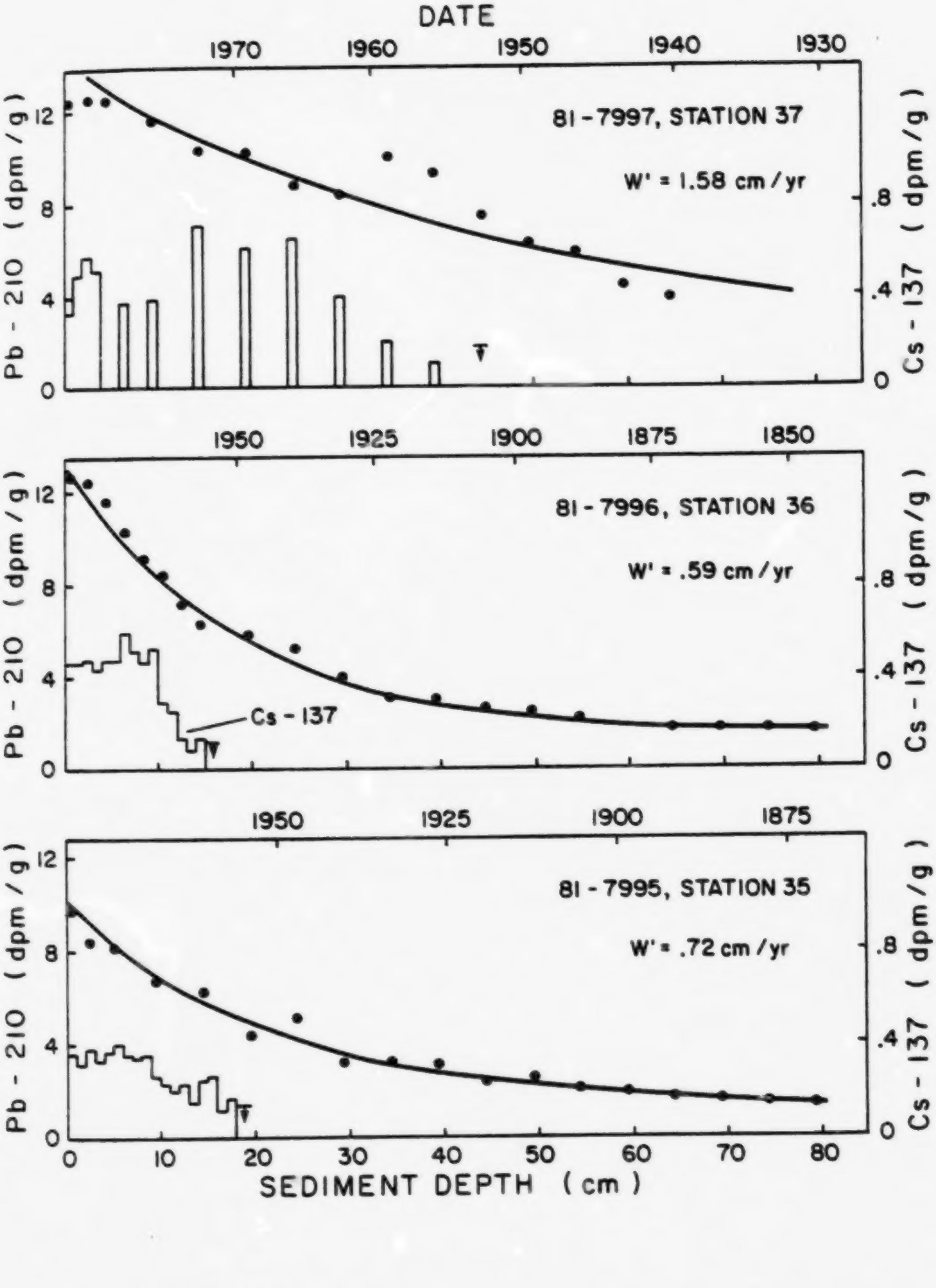


Figure 18. Pb-210 (circles) and Cs-137 (lower solid lines) profiles for cores from the Quoddy region (Figure 16). Apparent sedimentation rates (W') and deposition dates estimated from Pb-210 data.

conclusion is drawn from the observation in several cores that Cs-137 is measurable at sediment depths corresponding to dates of deposition prior to 1952 (estimated from the apparent deposition dates given in upper portion of each figure), which is the date for the initial introduction of Cs-137 in significant quantities into the environment. The apparent sedimentation rates estimated for these cores represent upper limits on actual sedimentation rates, as discussed above.

The apparent sedimentation rates increase sharply from a value of 0.18 cm/yr at station 30 to values of 0.72 cm/yr and 0.88 cm/yr at stations 35 and 27, respectively. This corresponds to a transition from the comparatively low sedimentation rate regime in the vicinity of Point Lepreau to a much higher depositional regime in the Quoddy region. A further increase in the apparent sedimentation rate is evident in the value of 1.58 cm/yr measured at Station 37, near the mouth of Passamaquoddy Bay which is one of the principal depositional sites for fine-grained material, as indicated in Figure 13. Increases in the apparent sedimentation rate in these regimes are also reflected by proportional increases in the total Pb-210 inventories in the sediments. This result indicates that the flux of Pb-210 to the sediments is mainly a function of the sedimentation rate and that particulate Pb-210 activities in the water column are similar throughout the western Bay of Fundy. Sediment-depth profiles for Cs-137 are also illustrated in Figure 18 for cores from the high sedimentation regime. The most interesting aspect of these data is that the threshold for Cs-137 occurs immediately above the sediment deposition date of 1952 for each core. The good agreement of the Cs-137 threshold with the deposition data for its initial input into the world's oceans indicates that there has been little sediment mixing in these regimes and that the "apparent" sedimentation rates represent the actual sedimentation rate at these locations.This result, if confirmed in the analysis of future cores, implies that a well preserved record of sediment inputs may be available at key locations in the Bay of Fundy. In this case the historical record of particle-reactive radionuclides released from the Point Lepreau NGS may be resolved at any future date through examination and dating of the sedimentary record. The value of Pb-210 dating in resolving the time dependence of pollutant inputs to the marine environment has been demonstrated by a study of the historical record of Hg releases from a chlor-alkali plant located in the Saguenay Fjord, Quebec (Smith and Loring, 1981).

The Cs-137 inventories in the three cores noted in Figure 18 are 4.1 dpm cm^{-2} (core 7995), 4.5 dpm cm^{-2} (core 7996) and approximately 11.2 dpm cm^{-2} (core 7997), respectively, all estimated using an average, measured, in situ, solid phase density of 0.7 g cc^{-1}. The integrated inventory of Cs-137 deposited at the latitude of New York City since the early 1950's as the result of nuclear weapons tests is approximately 26 dpm cm^{-2}, decay corrected to 1981. Cesium-137 inventories of this magnitude should be found in undisturbed soil columns. The considerably reduced Cs-137 inventories in the sediments of the Bay of Fundy indicate that most of the Cs-137 remains in solution in seawater and that only a small fraction of the fallout Cs-137 is sorbed onto particles and deposited in the sediments. In view of the fact that less than 10% of the Cs-137 in the water column is actually sorbed onto particles, the Cs-137 inventories in the Quoddy region are actually rather large, a result which further illustrates the efficiency of this sediment regime as a fine particle depositional sink.

The ratios of the Pb-210 inventory to the Cs-137 inventory in the three cores noted in Figure 18 are 39.4 (core 7995), 39.5 (core 7996) and 44.8 (core 7997). The similar values for relative inventories for these three cores, despite differences in their sedimentation rates, indicate that the same class of particles is being deposited at each location. In contrast, the relative Pb-210/Cs-137 inventories for the three cores in Figure 17 are 16.8 (core 7096), 37.5 (core 8098) and 54.4 (core 3514). This comparatively broad range of inventory ratios indicates that the nature of the sediment deposition and post-depositional processes may vary significantly on a transect through these different sediment regimes. In general, sediment regimes having high bioturbation rates compared to sedimentation rates also have reduced ratios of Pb-210/Cs-137 inventories. The greater influence of bioturbation in these regimes could lead to enhanced irrigation of sub-surface sediments through tubes and burrows which could result more efficient removal of Cs-137 from solution onto sediment particles. Alternatively, bioturbation may result in increased resuspension of sediments, followed by recycling of particles through the water column where they may augment their inventory of Cs-137 through surface sorption phenonema. Subsequent re-deposition of these particles may explain the net increased inventory of Cs-137 compared to Pb-210.

Releases of Cs-137 from the reactor based upon the past history of CANDU reactors will also be of the order of 0.1 Ci yr^{-1}. Since much of the reactor-released Cs-137 will likely remain in solution and be transported away from Point Lepreau, it is unlikely that significant quantities of reactor-derived Cs-137 will be detected above fallout levels of Cs-137 in most sediment regimes near Point Lepreau. However, isolated, fine-grained sediment regimes located in the generally sandy sediments near the reactor outfall may prove to be efficient depositional sites for more particle-reactive radionuclides such as Co-60 and these sites will be monitored subsequent to the start-up of the reactor.

3.2.3 Plutonium-239,240 and Pu-238 Results

Plutonium results for Bay of Fundy cores are also germane to the above discussion. Plutonium-239 and Pu-240 are particle-reactive fallout radionuclides whose atmospheric deposition record since 1962 has been proportional to that of Cs-137 (Thomas and Perkins, 1975; Harley, 1975). The mean fallout activity ratio of (Pu-239 + Pu-240)/Cs-137, estimated by Harley (1975) from measurements conducted on stratospheric particulate samples collected during the early 1960s is 0.012. Plutonium-239,240/Cs-137 ratios in soils tend to be close to fallout debris ratios (Hardy et al., 1973), while elevated ratios (0.01-0.05) have been measured in freshwater sediments (Edgington and Robbins, 1975a,b). In contrast, Pu-239,240/Cs-137 ratios in marine sediments can be greater by an order of magnitude (Livingston and Bowen, 1979) as a result of reduced particle/water distribution coefficients for Cs in higher ionic strength media such as seawater (Cs-137 must compete with seawater cations, especially potassium, for binding sites on particle surfaces).

Radionuclide data for two Bay of Fundy cores and one core collected from a freshwater lake (in the vicinity of Point Lepreau) are given in Table 6. Core 79-2770 was collected in a low sedimentation regime (same location as Sta. 23; Figure 2) and core 80-3437 was collected in a high sedimentation in the Quoddy region (Sta. 9; Figure 2). The Pb-210 and Cs-137 profiles for core 79-2770 were reported in Smith et al. (1981) and the Pb-210 profile for core 80-3437 is similar to that of core 81-7997

Table 6. Radionuclide results for several sediment depth intervals for two cores (#79-2770 and #80-3437) collected in Bay of Fundy and core #81-04636 collected in Retreat Lake near Point Lepreau.

Core 79-2770 (Sta. 3)

Core Depth (cm)	Pb-210 (dpm/g)	Cs-137 (dpm/g)	Pu-239,240 (dpm/kg)	Pu-238 (dpm/kg)	$\frac{\text{Pu-239,240}}{\text{Cs-137}}$	$\frac{\text{Pu-238}}{\text{Pu-239,240}}$
0-1	7.79 ± 0.37	0.32 ± 0.06	70 ± 3	3 ± 1	0.22	0.04
2-3	4.10 ± 0.25	0.19 ± 0.03	32 ± 1	2 ± 1	0.17	0.06
4-5	2.28 ± 0.17	<0.10	8 ± 1	<1.0	-	-

Core 80-3437 (Sta.9)

Core Depth (cm)	Pb-210 (dpm/g)	Cs-137 (dpm/g)	Pu-239,240 (dpm/kg)	Pu-238 (dpm/kg)	$\frac{\text{Pu-239,240}}{\text{Cs-137}}$	$\frac{\text{Pu-238}}{\text{Pu-239,240}}$
1-2	14.19 ± 0.45	0.54 ± 0.07	160 ± 20	7 ± 2	0.30	0.04
3-4	11.60 ± 0.35	0.46 ± 0.08	200 ± 20	5 ± 2	0.43	0.03
8-9	10.55 ± 0.62	0.44 ± 0.07	190 ± 10	6 ± 1	0.43	0.03

Core 81-04636 (Retreat Lake)

Core Depth (cm)	Pb-210 (dpm/g)	Cs-137 (dpm/g)	Pu-239,240 (dpm/kg)	Pu-238 (dpm/kg)	$\frac{\text{Pu-239,240}}{\text{Cs-137}}$	$\frac{\text{Pu-238}}{\text{Pu-239,240}}$
0-1	43.74 ± 2.40	38.95 ± 2.41	550 ± 80	15 ± 7	0.014	0.027 ± 0.013
4-5	27.41 ± 1.51	14.10 ± 1.10	470 ± 80	19 ± 7	0.033	0.040 ± 0.016
9-10	13.21 ± 0.58	1.47 ± 0.32	210 ± 20	7 ± 3	0.144	0.033 ± 0.015

(Figure 18) which was collected at the same location. Note that the Pu-239,240/Cs-137 ratios are considerably greater in the marine cores compared to the freshwater core, consistent with previous observations made regarding the comparative behaviour of plutonium and cesium in marine and aquatic systems. Note, also, that the Pu-239,240/Cs-137 ratios in core 80-3437 (high sedimentation regime) are twice as great as those in core 79-2770, collected in a low sedimentation rate regime where bioturbation rates are enhanced compared to sedimentation rates. Similarly, the inventory ratio of Pb-210/Cs-137 in core 79-2770 (=9.7) is considerably reduced compared to the ratios in high sedimentation regimes noted above. Again, these results may simply reflect differences in the nature and size distribution of particles deposited in the different sedimentation regimes. Alternatively, as noted previously, bioturbation may play a role in enhancing the inventory of Cs-137, compared to the more particle-reactive radionuclides, Pb-210 and Pu-239,240, in the low sedimentation regimes. The comparative uniformity in the Pu-239,240/Cs-137 ratios with depth in the sediment column reflects the similarities in the atmospheric input functions for these two radionuclides and indicates that, to a first approximation, selective remobilization or diffusion of either radionuclide with respect to the other is not occurring in these sediments to a significant extent.

Prior to 1964 the isotopic ratio, Pu-238/Pu-239,240 in nuclear weapons fallout debris was estimated to be 0.024, based on measurements conducted on a large number of soil samples collected in northern latitudes (Hardy et al., 1973). In April, 1964 a navigational satellite carrying a SNAP-9A nuclear power generator, containing 17 kCi of Pu-238, was burned up upon re-entry into the southern hemisphere stratosphere, resulting in a tripling of the global inventory of Pu-238 by 1970. SNAP-9A derived Pu-238 was first observed in surface air particulate material in 1966 at Richland, Washington (Thomas and Perkins, 1975) and since that time the isotope ratio Pu-238/Pu-239,240 has varied in different environmental phases in a complex manner (Smith and Ellis, 1982). In general, the isotope ratios of these radionuclides lies in the range of 0.03-0.15 in any sediments containing recently-deposited (post-1966) material. Values of this ratio measured in the range 0.03-0.06 indicate that some component of SNAP-9A-derived Pu-238 has been deposited in Bay of Fundy sediments, a result consistent with the comparatively recent deposition (and mixing) of this material. Measurements of this ratio provide an important quality control check on the experimental plutonium results and, in addition, Pu-238 measurements will be used to provide geochronological information in sediment deposition in Future Bay of Fundy studies.

3.3 Laboratory Intercomparison Program

The AERU laboratory participates in an intercomparison program carried out by the Environmental Protection Agency (EPA, Las Vegas, Nevada) in order to maintain a check on the analytical precision and accuracy of the laboratory's radionuclide results. Water samples and air filters containing known concentrations of radionuclides are shipped to radioactivity laboratories throughout the U.S. and Canada and a report containing the results of each intercomparison experiment is subsequently issued by the U.S. EPA. The AERU results for 1981-82 are listed in Table 7, while results for the period 1979-1980 are given in Smith et al., 1981. Postal delays resulted in some of the AERU results (denoted by an asterisk) not

Table 7. Results of AERU laboratory intercomparison experiments with the Environmental Protection Agency (Las Vegas, Nev.). Asterick denotes results not included in EPA report.

Sample Matrix	Collection Date	Nuclide	EPA Value	AERU Value	Grand mean (All Labs)	2 SD (AERU Expt.)	2 SD (All Labs)	AERU/EPA
Water	13/2/81	H-3	1760 pCi/l	1760 ± 98 pCi/l	1778 pCi/l	20	460	1.00
Water	12/2/81	H-3	1950 pCi/l	1930 ± 78 pCi/l	1948 pCi/l	40	484	0.99
Water	9/10/81	H-3	2210 pCi/l	2277 ± 60 pCi/l	2133 pCi/l	42	428	1.03
Water	11/12/81	H-3	2700 pCi/l	2630 ± 85 pCi/l	2676 pCi/l	124	448	0.97
*Air Filter	19/12/80	Cs-137	19 pCi	19.7 ± pCi/filter	24 pCi	-	12	1.04
Air Filter	24/3/81	Cs-137	14 pCi	14.9 ± pCi/filter	16 pCi	4	8	1.06
Air Filter	25/7/81	Cs-137	19 pCi	20.5 ± pCi/filter	24 pCi	2	12	1.08
Air Filter	25/7/81	Cs-137	19 pCi	20.4 ± pCi/filter	24 pCi	-	12	1.07
Air Filter	26/3/82	Cs-137	23 pCi	23.3 ± pCi/filter	25 pCi	2	14	1.01
Water	6/2/81	Co-60	25 pCi/l	25 ± 1 pCi/l	25 pCi/l	2	8	1.00
Water	6/2/81	Zn-65	85 pCi/l	47 ± 7 pCi/l	89 pCi/l	10	22	.53
Water	6/2/81	Cs-134	36 pCi/l	33 ± 3 pCi/l	33 pCi/l	6	10	.92
*Water	6/2/81	Cs-137	4 pCi/l	4 ± 1 pCi/l	5 pCi/l	-	4	1.0
*Water	9/10/81	Cs-137	15 pCi/l	16 ± 1 pCi/l	16 pCi/l	-	6	1.07
*Water	4/12/81	I-131	76 pCi/l	75.3 ± pCi/l	69 pCi/l	-	20	1.01
Water	5/2/81	Cs-137	23 pCi/l	18.3 ± 1 pCi/l	24 pCi/l	2	8	1.25
Water	2/4/82	I-131	62 pCi/l	52 ± 6 pCi/l	63 pCi/l	12	16	1.18

being included in the final EPA report. For each intercomparison experiment (with the exception of the Zn-65 measurement in water) the AERU measurements was within the analytical uncertainty (2 standard deviations) of the EPA reported value. The large error in the Zn-65 measurement was unusual because the Co-60 measurement, performed by analysis of the same gamma spectrum of the same environmental water sample, was in excellent agreement with the EPA result. The sample was subsequently analysed for Zn-65 for a second time and the result was in better agreement with the EPA result (Table 7). The experimental standard deviation for the AERU results, usually based on three determinations, was in each case acceptable by EPA standards. These results indicate that there are no apparent precision or accuracy problems with AERU results for the analyses of radionuclides distributed in these experiments.

4.0 ATMOSPHERIC MONITORING

The atmosphere is the most dynamic environmental medium available for the transport of gaseous and particulate releases from nuclear power plants. Radioactive isotopes will be released in stack gases, transported over land and sea and subsequently exchanged with surface water, incorporated into terrigenous media or precipitated in rain, snow or dry particulate fallout. The two principal components of the atmospheric portion of the Point Lepreau environmental monitoring program are (1) compiling a meteorological data base for the Point Lepreau region which can be applied in projections of the diffusion and dispersion of airborne pollutants under diverse weather conditions and in atmospheric transport models, and (2) the collection of atmospheric water vapour, and particulate samples for radioactivity analyses.

4.1 Meteorological Data Collection

A meteorological station was established in 1978 at Hall's Lake Siding approximately 4 miles north-northeast of the Point Lepreau Nuclear Generating Plant. This station was equipped with (1) U2A wind equipment to yield wind speed and direction at 10 metres (above ground level) in chart form, (2) a Fischer and Porter precipitation gauge, and (3) a thermohydrograph to record in chart form, temperature and relative humidity. The criteria which determined the choice of Hall's Lake Siding as a meteorological station were (1) lack of obstructions which might interfere with the wind flow or generate random noise in the precipitation, temperature or relative humidity measurements, (2) accessibility (the meteorological station is just off the Saint John-Saint Stephen Highway), and (3) availability of personnel to routinely check and maintain the equipment.

The meteorological data collected at Hall's Lake Siding provides important baseline data characterizing meteorological conditions throughout the year in the vicinity of Point Lepreau. A summary of the wind speed and direction distributions is given in Table 8. There is no doubt that wind speeds in coastal areas exceed those in areas more remote from the water. Surface roughness differences between land surfaces and adjacent open water account for much of this discrepancy. Additionally, it is well known that east coast storms tend to track south of land surfaces, thus causing a gradient of wind speeds. Locations farther north area, on the average, subject to lighter winds than those closer to the storm centre. This decrease in wind speed as one moves inland from the open water may take place quite abruptly.

There is evidence that the wind speeds registered too low on the wind speed sensor during 1979-1981 consequently, the recorder associated with the sensing equipment has been replaced. However, it is believed that the wind direction distributions indicated in Table 8 are quite reliable. There is a marked preference for a southwesterly wind direction which is especially pronounced during the warmer months. Mini-sonde evidence was presented in a previous report (Smith et al., 1981) to suggest that the summer period was characterized by low atmospheric mixing heights and low ventilation coefficients, conditions which would lead to the inefficient dispersion of atmospheric pollutants and radionuclides. The wind direction data indicate that there is a strong probability that airborne materials, radioactive and otherwise, emitted at the Point Lepreau NGS site will be transported inland. Since the summer period of maximized southwesterly

Table 8. **Percent frequency distributions of windspeeds and direction from U2A Wind Sensor at Hall's Lake Siding during period March 1979 to August 1981. Windspeeds in knots.**

	Period June-Sept.		Period Dec. Jan. Feb. Mar.		Annual	
Direction	% Frequency	Mean Speed	% Frequency	Mean Speed	% Frequency	Mean Speed
N	1.7	4.4	4.1	8.4	2.8	6.3
NNE	1.4	4.5	3.5	6.1	2.5	5.9
NE	1.2	2.5	2.1	3.2	1.6	3.2
ENE	1.5	2.8	2.1	4.1	1.8	3.4
E	3.0	2.5	7.3	7.1	5.6	5.3
ESE	3.7	3.8	6.2	6.1	5.5	4.8
SE	3.6	4.0	6.2	4.6	4.6	4.4
SSE	6.6	4.2	3.7	7.6	5.8	4.8
S	14.9	4.6	7.9	7.3	12.1	5.4
SSW	21.9	4.3	8.6	7.0	16.0	5.0
SW	21.0	4.1	10.6	6.2	15.7	4.7
WSW	5.4	4.2	11.2	6.6	7.0	5.5
W	4.8	5.0	9.6	6.6	6.6	5.6
WNW	3.0	6.1	6.6	6.9	5.0	6.9
NW	3.6	7.1	7.6	8.8	4.9	8.5
NNW	2.4	6.0	2.5	7.4	2.2	6.7

wind direction also coincides with the period of least efficient atmospheric dispersion, there is an increased probability that materials emitted from the plant during the summer months will be transported and deposited along a northeasterly trajectory extending from the plant site. Consequently, it is anticipated that terrestrial monitoring results from regions northeast of the plant site may exhibit the greatest radionuclide concentrations due to the presence of the reactor.

Cold, winter-time, northwesterly air flows are generally warmed over the Bay of Fundy. The net effect is to raise the atmospheric mixing height and to disperse airborne materials in turbulent air motions which act vertically as well as laterally. Therefore, radionuclides emitted from the reactor under winter-time conditions of northwesterly winds will probably be more efficiently dispersed in the atmosphere during transit across the Bay of Fundy. It may be concluded from the meterological data that deposition of airborne radionuclides emitted from the Point Lepreau NGS will be maximized during the summer months on a transect extending in a northeasterly direction from the plant site.

4.2 Air Monitoring Stations

Four air monitoring stations, described in Bishop et al. (1980) and illustrated schematically in Figure 19 have been deployed in the Lepreau region at the sites noted in Figure 20. Three of these stations are operated by local residents, the fourth by the New Brunswick Electric Power Commissions's Nuclear Operations Group. Two of the AERU stations (Point Lepreau and Dipper Harbour) are located near similar stations owned and operated by NBEPC and since their sampling periods are being coordinated their data will be useful for comparison purposes. A fifth station

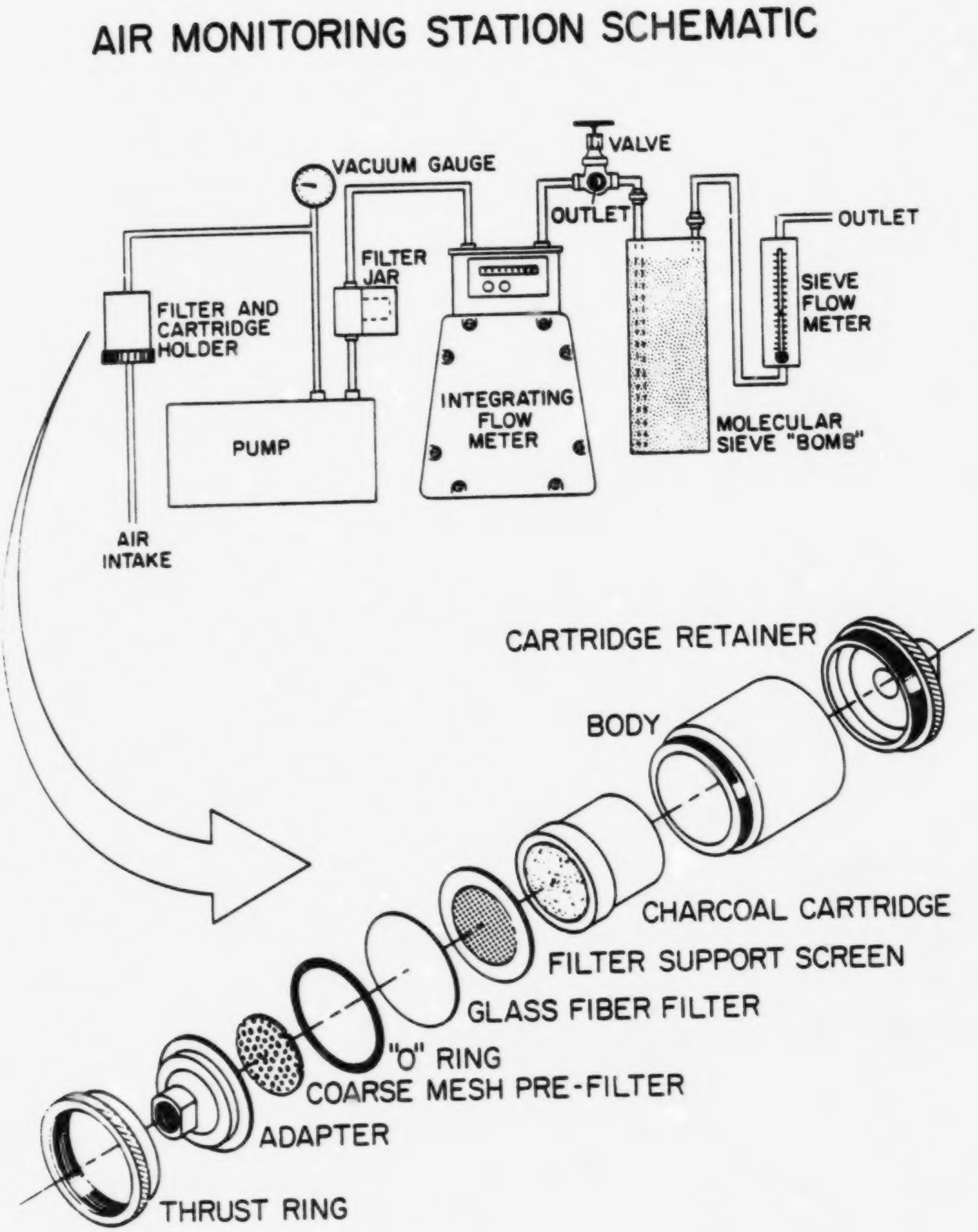


Figure 19. Schematic diagram of air monitoring station used in the field to collect water vapour (on molecular sieve) particulates (glass fiber filter) and I-131 (on charcoal cartridge).

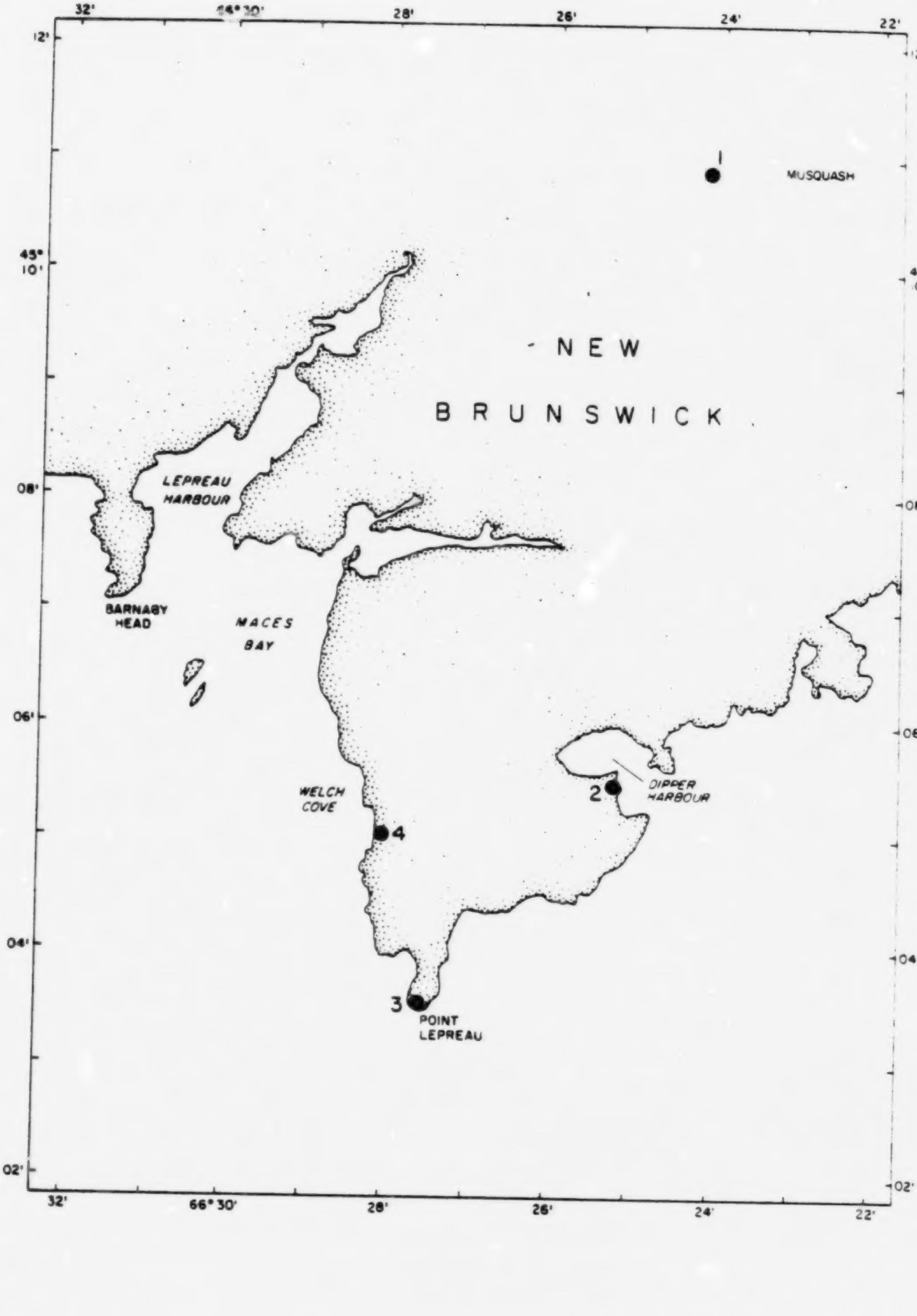


Figure 20. Location of four air monitoring stations in vicinity of Point Lepreau. Fifth station is located at Digby, N.S.

also operated by a local resident was established at Digby, N.S. in 1981, in order to determine the extent of radionuclide transport from the Point Lepreau NGS across the Bay of Fundy.

During the pre-operational period the air monitoring stations are operated for approximately one week of each month. Air is continuously drawn into the system at a height of 1 meter above the ground via steel piping that extends vertically through the bottom of the sampler housing. Thus, the collection of wet and dry precipitation is minimized except under the most adverse meteorological conditions.

The 1/4 horsepower, oil-less vacuum pump draws air through a 47 mm diameter type "A" glass fibre filter and a cartridge of activated coconut shell carbon supplier. The filter removes 0.3 µm particles from the air stream with a retention efficiency of 99% and the charcoal bed adsorbs elemental and organically bound forms of iodine with a retention efficiency of 93%. A modification was made to the arrangement of the filters and cartridges in their holders at all stations in July 1981. To prevent the uneven accumulation of particulates on the filter caused by adhesion of the coarse mesh pre-filter from adhering to the glass fiber filter, the retaining assembly indicated in Figure 19 was inverted and the internal components rearranged. The glass fiber filter now rests on the "O" ring with good separation from the pre-filter and air is drawn through the cap of the charcoal cartridge. Flow rates, determined by timing the passage of a known volume of air through the integrating flow meter, drop during sampling because of particle accumulation and consequent filter clogging. This drop in the flow rate often results in a lowering of the flow rate through the molecular sieve "bomb", used to remove water vapour from a diverted portion of the filtered air stream. However, from the data supplied by the operators, it is possible to compensate for these flow rate changes.

Sieve air flow rates are regulated with a calibrated flow meter connected to the cannister outlet and are adjusted according to estimates of seasonal variation in absolute humidity. This ensures that the water sample collected is within the sieve's retention capacity. Since relative humidity and temperature data are available for the Musquash site (Station 1), from the meteorological station deployed at the same location, the absolute humidities during the air sampling periods may be calculated. These are compared with sieve water recovery data as an additional check on the system's performance.

At the end of an air sampling period, the operators return the particle filters, charcoal cartridges and molecular sieve cannisters for radionuclide analyses. The approximate detection limits, which vary depending on sampling and instrumental conditions, are $4x10^{-5}$ Bqm^{-3} for particulate Cs-137, $3x10^{-5}$ Bqm^{-3} for I-131 on the charcoal cartridge and 1 to $7x10^{-2}$ Bqm^{-3} for H-3 in water vapour. These L_D's assume a decay interval between the midpoints of the sampling and analysis intervals of 10 days.

In Bishop et al. (1980) it was noted that when the mean tritium in seawater activities were calculated for data sets that included "less than" values, all net count rate data was used, including negative cpm numbers. This procedure was followed in order to avoid introducing a bias in favour of the higher activity concentrations that would have been caused if the L_T numbers had been dropped from the data sets. This was possible due to that fact that the same counting conditions applied for all the samples, resulting in a narrow range of counting efficiencies, and which permitted the use of an average efficiency to arrive at mean seawater H-3 activities.

However, the liquid scintillation counting conditions were varied during the course of 1981 to take advantage of the availability of larger capacity counting vials and in order to improve on the resistance of the counting matrix (gel) to laboratory temperature fluctuations. The result of this is that due to varying water to cocktail ratios and counting efficiencies, the net count rate data cannot be used to obtain means of data sets. Therefore, tritium activities in atmospheric vapour samples were calculated for *all* net count rates, including negative ones, each with its particular counting efficiency. Obviously, activities calculated for "less than" samples are not real since the count rates do not differ significantly from their backgrounds (at the 95% confidence level). Negative values (in Bq l^{-1}) cannot be assigned zero activity since this again bias the data set in favour of the higher numbers due to the inclusion of positive data points that also may not differ significantly from their backgrounds.

4.2.1 Tritium results

All air monitoring stations have been operating satisfactorily with individual sampling periods of 6 to 8 days during which 400 to 1110 cubic meters of air were drawn through the filters. This variation in sampling volume is due to varying particulate loads in the air and varying sampling periods. Sampling was temporarily suspended at the four New Brunswick stations in October 1981, while their electrical circuitry was being inspected.

The molecular sieve cannisters are returned to the AERU laboratory from the field where water vapour is removed from the molecular sieve material by heating at 200°C under vacuum. The distillate was collected by freezing in a refrigerated trap held at -10°C. This method was used for all samples collected up to and including the month of August 1981, in order to minimize the samples' exposure to laboratory air which, up to that point in time had been contaminated with tritium from a radioisotope storage safe. Unfortunately, handling of the hygroscopic sieve under these conditions did result in some contamination which resulted in the loss of 5 of the 39 samples collected in 1981. After removal of the offending safe and the return to acceptable laboratory air tritium levels, distillations were performed at 350°C under atmospheric pressure. The distillate was collected in a refrigerated trap held at -20°C and was protected from laboratory air by a moisture trap at the system outlet.

The results of tritium analyses on samples of atmospheric moisture are given in Figure 21. The error bars represent the 2 sigma counting uncertainty. There is no apparent seasonal bias in the specific tritium activity in atmospheric water vapour. Thus, there is no evidence from these results for enrichment due to stratospheric inputs of tritium during the spring and summer months or to increased inputs of tritium to the atmosphere due to recent nuclear weapons tests conducted by the People's Republic of China. During this period of time, tritium analyses were performed on precipitation samples collected at the Musquash meteorological station by AES. Good agreement was observed between tritium results for these samples and results from atmospheric moisture collected at the air monitoring station indicating that rainfall neither enhanced nor diluted airborne tritium activities to a measurable extent. The mean specific tritium activity measured during 1981 was 5.9 ± 3.8 Bq l^{-1}.

Both the volume of air passed through the molecular sieves and the total weight of atmospheric water vapour (collected with an efficiency of close to 100%) retained in the sieve are measured for each sampling run.

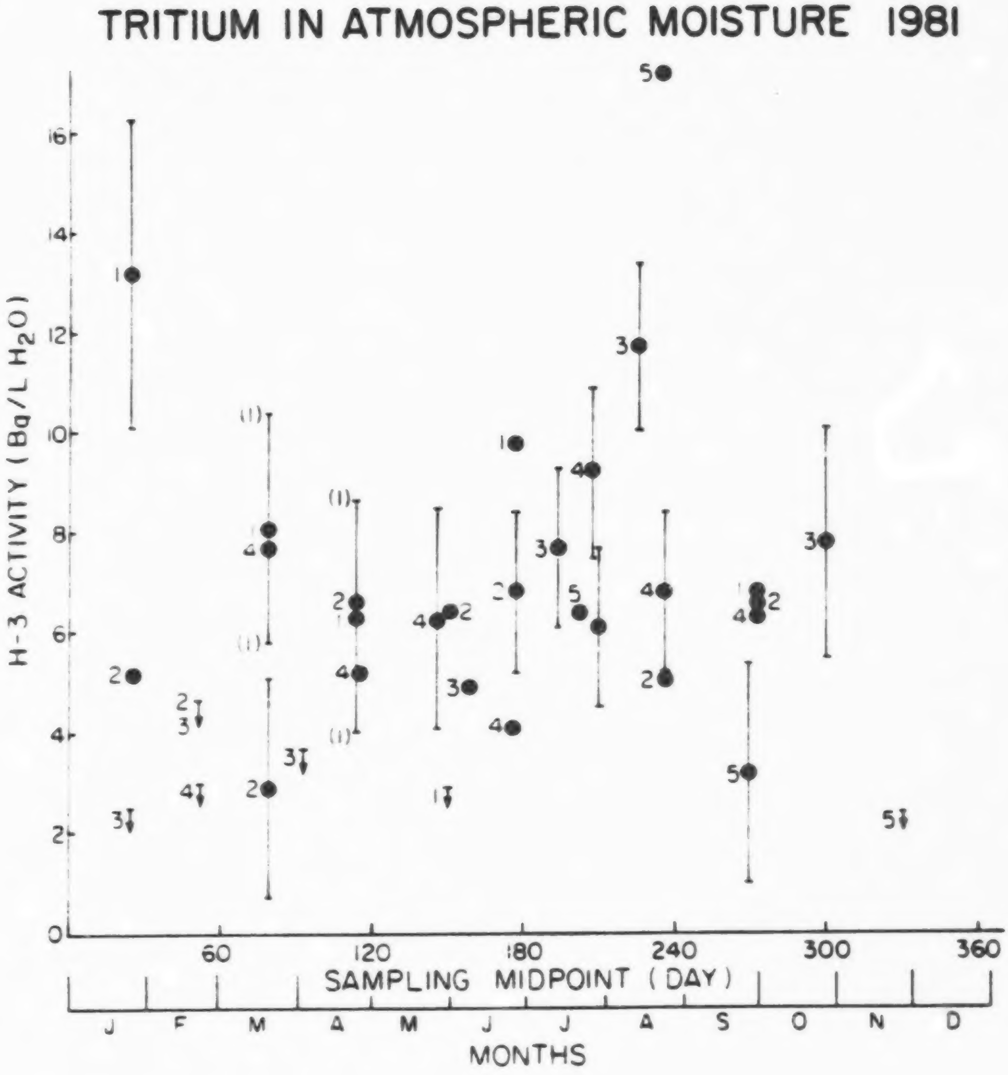


Figure 21. Tritium measurements for different air monitoring stations (identified by numbers; Fig. 20) as function of sampling date. Arrows indicate value below the detection limit.

Hence, the tritium activity in terms of air volume can be estimated as a function of air volume and the integrated absolute humidity can be determined for each sampling period. These parameters are plotted in Figures 22 and 23, error bars being given where possible (to indicate the uncertainty associated with each measurement). The theoretical absolute humidities at 100% relative humidity and air temperatures of 0°C, 10°C, 30°C, and 30°C are also indicated in Figure 23. Clearly, the very pronounced summer peak in the tritium activity in air is due to the increased humidity during this time of the year. These results indicate that tritium is transported into the study area with water vapour and that there is no apparent source term for tritium that operates independently of the continent-ocean-atmosphere hydrological cycle. The absence of a seasonal bias in baseline levels of tritium in atmospheric water vapour should simplify the procedure for identifying releases of tritium from the Point Lepreau NGS.

4.2.2 Radionuclides in atmospheric particulates

Air filters containing the airborne particulates were counted for gamma emitting radionuclides on hyper-pure Ge or Ge(Li) detectors. Air filter results for 1980 and 1981 are given in Table 9. The station numbers correspond to locations given in Figure 20, with station 6 located at the Bedford Institute of Oceanography (Dartmouth, N.S) in 1980. During the first five months of 1981, air filter samples from several stations were combined and analyzed together, as indicated in Table 9, in which case the sampling date corresponds to the approximated mid-point of their combined sampling periods and the air volume is the sum of the air volumes passed through each station.

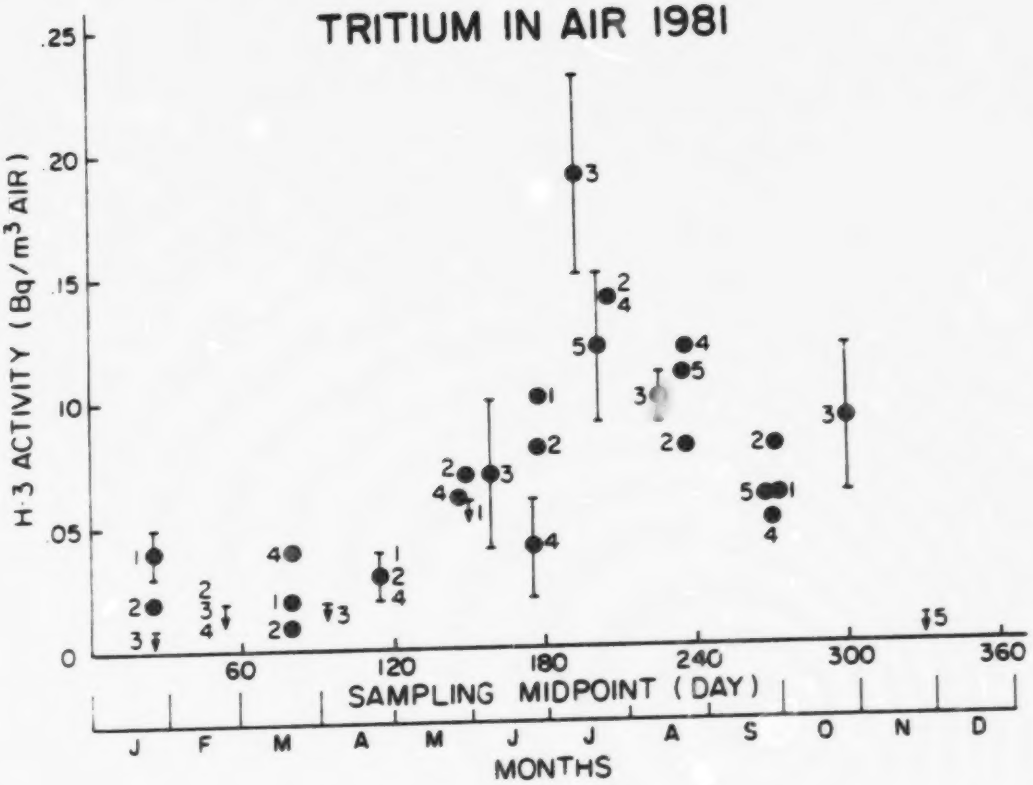


Figure 22. Tritium activities as function of air volume sampled for different sampling intervals. Summer peak in tritium is caused by enhanced atmospheric moisture content (Fig. 23).

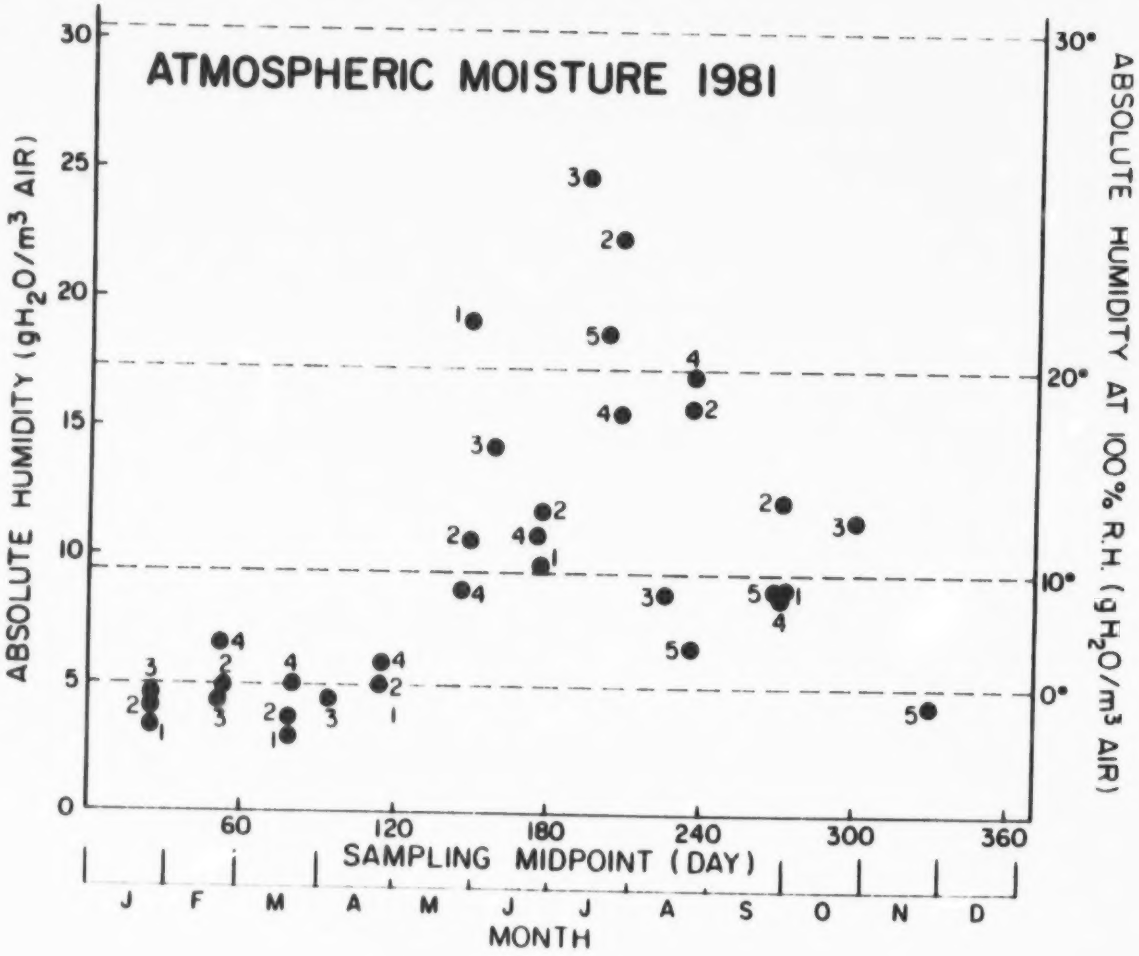


Figure 23. Absolute humidity for the sampling periods corresponding to tritium measurements given in previous figures. Dashed lines correspond to theoretical absolute humidities at the different temperatures.

the different temperatures.

Table 9. Radionuclide results for air particulate material collected on air filters at air monitoring stations identified by location number in Figure 20. Stations 5 and 6 were located at Digby, N.S. and Dartmouth, N.S., respectively. Date denotes midpoint of sampling interval which usually lasted 6-8 days. Radionuclides not listed were generally below the detection limit. Nb-95 cannot be precisely measured when Zr-95 is below the detection limit.

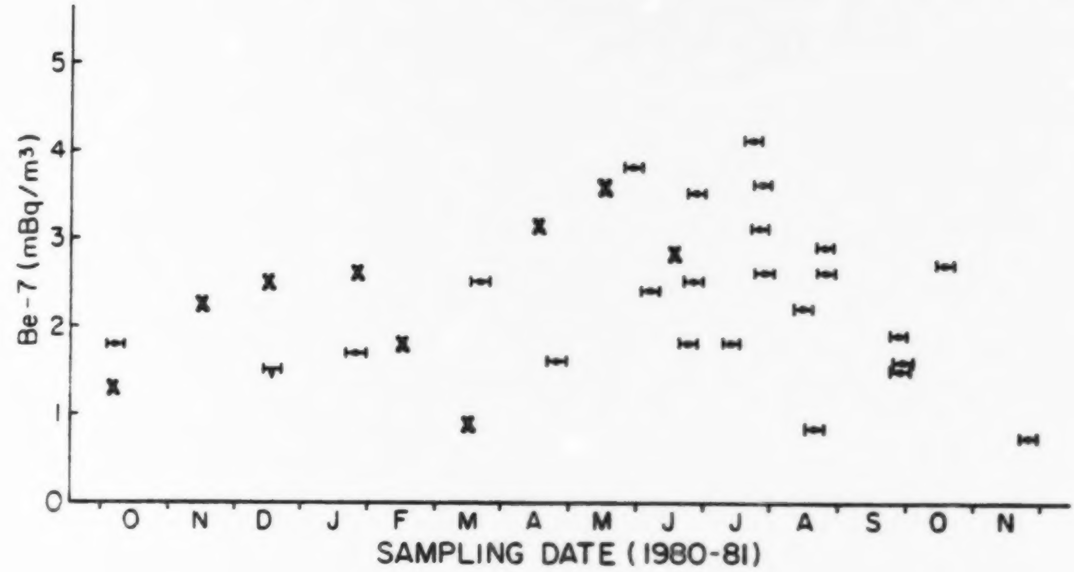
Sample						Radionuclide Activities (mBq/m^3)					
No.	Station	Date	Volume (m^3)	Cs-137	Be-7	Ce-141	Ce-144	Ru-103	Ru-106	Zr-95	Nb-95
807566-8	2,3,6	5/10/80	2915	<0.01	1.8 + 0.2	<0.02	<0.05	<0.014	<0.12	<0.02	-
807574	6	21/10/80	112	<0.7	<6.1	<0.7	<2.9	<0.6	<6.7	<1.3	-
807589	6	17/12/80	766	<0.12	<1.5	<0.3	0.11 ± 0.06	<0.19	<1.2	<0.28	-
810601	1,2,3,4	25/1/81	3532	<0.01	1.7 ± 0.1	2.5 ± 0.01	0.16 ± 0.04	0.34 ± 0.01	<0.1	0.44 ± 0.02	0.57 ± 0.03
810610,1,3	1,2,4	20/3/81	2343	<0.02	2.5 ± 0.4	0.35 ± 0.04	0.66 ± 0.14	0.51 ± 0.05	<0.4	0.83 ± 0.08	1.69 ± 0.09
810615-8	1,2,3,4	24/4/81	2734	<0.03	1.6 ± 0.3	0.22 ± 0.04	0.81 ± 0.10	0.31 ± 0.04	<0.3	0.85 ± 0.07	1.55 ± 0.15
810619,20,2	1,2,4	28/5/81	2063	0.12 ± 0.03	3.8 ± 0.1	0.22 ± 0.04	1.6 ± 0.1	0.41 ± 0.04	0.9 ± 0.3	1.16 ± 0.08	2.15 ± 0.12
810621	3	6/6/81	852	<0.13	2.4 ± 0.7	0.20 ± 0.08	1.0 ± 0.3	0.17 ± 0.05	<1.1	0.78 ± 0.14	1.45 ± 0.17
810625	4	23/6/81	561	<0.12	1.8 ± 0.9	0.16 ± 0.04	0.63 ± 0.16	0.15 ± 0.05	<1.0	0.46 ± 0.15	0.91 ± 0.14
810623	1	25/6/81	750	0.21 ± 0.09	2.5 ± 0.7	<0.10	0.61 ± 0.14	0.13 ± 0.04	<0.8	0.32 ± 0.10	0.80 ± 0.16
810624	2	26/6/81	694	0.07 ± 0.03	3.5 ± 0.9	0.16 ± 0.09	0.77 ± 0.19	0.17 ± 0.10	0.5 ± 0.3	0.44 ± 0.17	1.24 ± 0.29
810626	3	12/7/81	665	<0.13	1.8 ± 0.6	<0.3	0.23 ± 0.08	<0.18	<1.3	0.34 ± 0.12	0.60 ± 0.24
810614	5	21/7/81	673	0.11 ± 0.04	4.1 ± 0.9	<0.2	0.95 ± 0.26	<0.14	<0.8	0.41 ± 0.15	0.90 ± 0.21
810629	4	25/7/81	581	0.05 ± 0.03	3.1 ± 0.7	0.07 ± 0.04	0.71 ± 0.24	0.09 ± 0.04	<0.65	0.47 ± 0.11	0.71 ± 0.12
810628	2	26/7/81	518	0.11 ± 0.04	3.6 ± 0.7	0.09 ± 0.04	0.98 ± 0.22	0.12 ± 0.04	0.4 ± 0.2	0.46 ± 0.13	0.79 ± 0.18
810627	1	27/7/81	685	0.05 ± 0.02	2.6 ± 0.6	<0.09	0.61 ± 0.22	0.05 ± 0.02	<0.6	0.30 ± 0.10	0.59 ± 0.14
810630	3	14/8/81	733	<0.08	2.2 ± 0.3	<0.1	0.56 ± 0.23	<0.09	<0.7	0.10 ± 0.06	0.19 ± 0.11
810631	1	19/8/81	320	<0.12	0.8 ± 0.4	0.09 ± 0.06	<0.5	<0.15	<1.0	<0.25	-
810634	5	22/8/81	449	0.07 ± 0.03	5.5 ± 0.8	0.13 ± 0.05	0.75 ± 0.30	<0.11	<0.8	0.20 ± 0.08	0.43 ± 0.15
810632	2	24/8/81	479	<0.14	2.9 ± 1.2	<0.19	0.3 ± 0.2	<0.18	<1.2	<0.31	-
810633	4	24/8/81	600	0.04 ± 0.03	2.6 ± 0.6	<0.09	0.32 ± 0.09	<0.09	<0.6	0.16 ± 0.06	0.32 ± 0.11
810636	2	27/9/81	620	<0.10	1.5 ± 0.7	<0.12	<0.38	<0.10	<0.8	<0.18	-
810638	4	27/9/81	841	<0.06	1.9 ± 0.5	<0.08	<0.25	<0.08	<0.5	<0.13	-
810635	1	29/9/81	801	<0.05	1.6 ± 0.6	<0.09	<0.29	<0.08	<0.7	<0.15	-
810642	3	27/10/81	694	<0.13	2.6 ± 1.1	<0.32	<0.80	<0.15	<1.1	<0.24	-
810649	5	25/11/81	1110	<0.06	0.8 ± 0.4	<0.2	<0.04	<0.06	<0.5	<0.10	-

With the exception of Be-7, which is a cosmogenic tracer, the radionuclides listed in Table 9 are produced as the result of nuclear weapons tests. Atmospheric radionuclide levels of fission products during 1980 were generally below the detection limit. On October 16, 1980 the People's Republic of China carried out a nuclear weapons test of approximately 200 Mt size at the Lop Nor test site, which resulted in a sharp increase in atmospheric radionuclide levels in the early part of 1981. This event provided an excellent opportunity to field test the air monitoring stations, as outlined below.

Beryllium-7

The rate of production of Be-7 by cosmic ray fragmentation of atmospheric nitrogen and oxygen in the upper stratosphere is exceeded only by that of C-14 and H-3. Its high specific activity, resulting both from its comparatively short half life of 53 days, and the lack of dilution by stable beryllium, make this an effective tracer in the study of atmospheric mixing and circulation (Lal and Peters, 1967). Furthermore, inputs of Be-7 as the result of nuclear testing have been generally undetectable in ground level air (Gustafson et al., 1961) although, recently, Be-7 from the testing of French nuclear devices has been detected in the stratosphere (Sagar and Goel, 1980). Because of its wide distribution in the atmosphere and its particle-reactive characteristics, Be-7 has proven useful as a particle tracer for rapid processes (on the order of months) in the marine environment and has also been applied to studies of sediment reworking processes in aquatic systems (Krishnaswami et al., 1980).

Beryllium-7 results for 1980 and 1981 for the AERU stations are illustrated in Figure 24. There is some indication of elevated levels



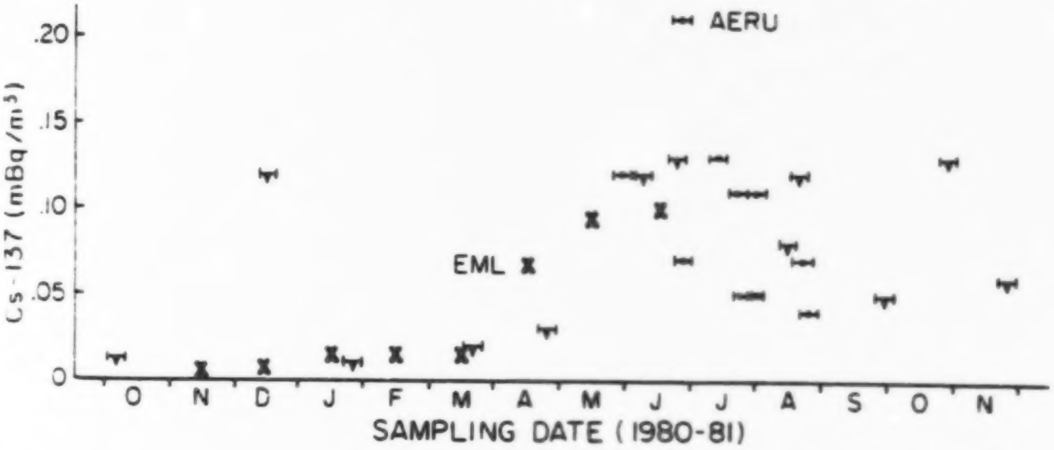


Figure 24. (Upper figure) Be-7 activity of air particulate material for 1980-1981. Arrows correspond to values below the detection limit. Crosses indicate values measured by Environmental Measurements Laboratory (EML) at Moosonee, Ont. (Lower figure) Cs-137 activity of air particulate material collected near Point Lepreau. Enhanced values in summer of 1981 due to Chinese nuclear weapons test, Oct. 1980.

during the summer of 1981, probably due to the unusual increase in the Be-7 concentration which occurs during the spring and summer months of each year (Gustafason et al., 1961). Beryllium-7 is routinely measured in surface air at a number of locations throughout the world by the Environmental Measurements Laboratory (EML) located in New York City. EML measurements of Be-7 (through June 1981) for a station in Moosonee, Ontario (latitude 51°16'N) are in reasonable agreement with the AERU results (Figure 24) despite the considerable month to month variability in both sets of data. Although there is some evidence of elevated levels of Be-7 measured at station 5 (Digby, N.S.) and slightly decreased levels at station 1 (Musquash), further monitoring is required to determine whether this trend is representative of atmospheric circulation within this region. Measurements of the naturally-occurring radionuclide, Be-7, provide a quality control check on the air monitoring results and may also prove useful in determining the influence of meterological phenonema (heavy rainfall, etc.) on atmospheric radionuclide levels.

Cesium-137

This history of nuclear weapons testing is clearly reflected in the historical record of Sr-90 deposition for New York City (Environmental Measurements Laboratory, 1982) illustrated in Figure 25. Since the half-life of Cs-137 (30.2 yr) is close to that of Sr-90 (28.8 yr) and both radionuclides are produced in similar quantities during nuclear weapons tests with a ratio of (Cs-137/Sr-90 = 1.5) then the historical record of Cs-137 deposition can be determined by multiplying the Sr-90 data in Figure 25 by a factor of 1.5.

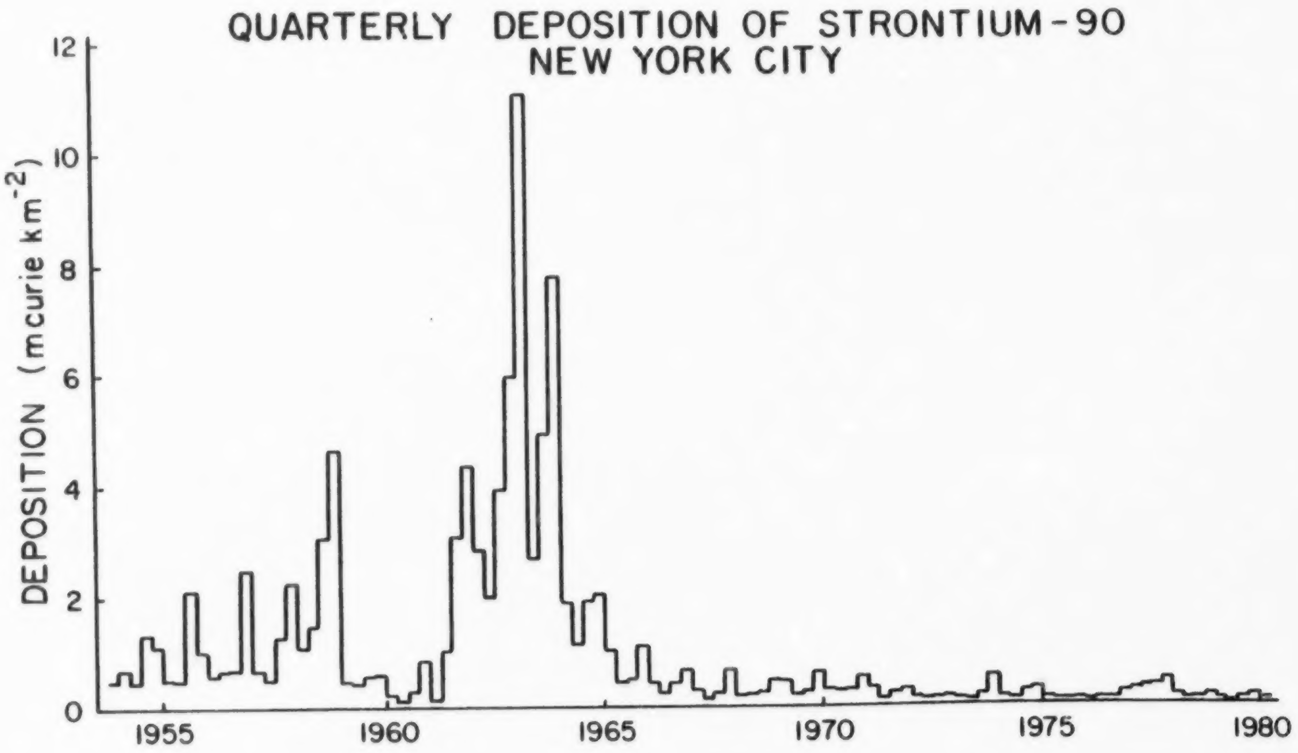


Figure 25. Quarterly fallout record of Sr-90 for New York City measured by the Environmental Measurements Laboratory (1982).

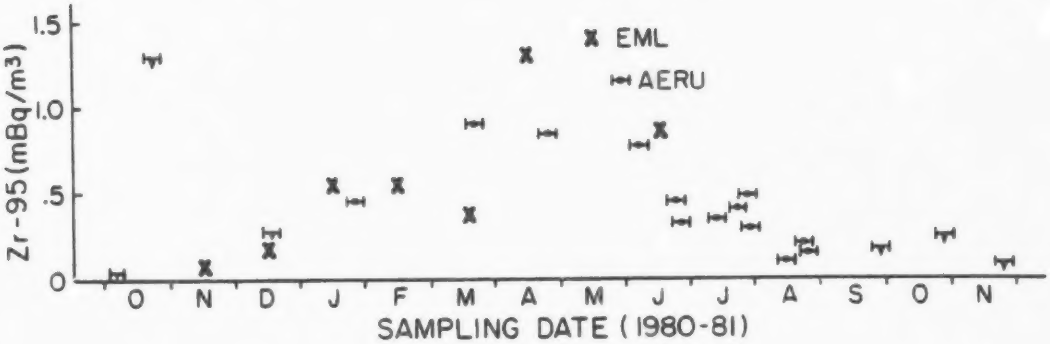
The first major nuclear weapons tests were conducted by the USA at test sites in the Pacific Ocean in 1952 and subsequently, monitoring of Sr-90 deposition was begun by EML in 1954. Testing by the USA and USSR during the 1950's produced a maximum in the Sr-90 inventory in the atmosphere in 1958-59 evident in the Sr-90 deposition record for New York City (Figure 25). A moratorium on nuclear testing between November 1958 and September 1961 led to a pronounced decrease in atmospheric radioactivity. The resumption of testing in 1961 followed by the introduction of the Limited Test Ban Treaty in October 1963 produced a second maxima in the historical record of Sr-90 deposition (Carter and Moghissi, 1977). This characteristic, double peak in the fallout record has been used to determine the geochronology for ice accumulation in Antartica (Koide et al., 1979) and Greenland (Koide et al., 1977) and for sediment accumulation in lakes (Edgington, 1981) and marine environments (Smith and Walton, 1980). Fallout deposition has declined since the 1960's with smaller peaks in the 1970's occurring as the result of tests carried out by India, France, England and the People's Republic of China. Much of the bomb produced radioactivity is initially injected into the stratosphere where the residence time of particle reactive radionuclides is of the order of 1 to 2 years compared to a tropospheric residence time of the order of months. Enhanced mixing of stratospheric and tropospheric air masses during the spring and summer months produces the yearly maxima evident in the fallout record.

In contrast to the Be-7 data, Cs-137 activities in surface air at Point Lepreau were below the detection limit in 1980 and then exhibited a sharp increase during the spring of 1981 (Figure 24). The comparatively high levels maintained during the summer of 1981 were, in part, due to inputs from the Chinese weapons test of October 1980. The agreement between the AERU results and EML results for the same period of time (Moosonee, Ontario) is reasonably good, particularly in view of the differences in meterological parameters between Moosonee and Point Lepreau, the latter site located 5° further south. The much lower detection limit for Cs-137 for the EML data is principally due to the larger volumes of air (≈40,000 m^3) pumped through their air monitoring stations each month compared to ≈1000 m^3 for the AERU stations. Clearly, it will be essential during the operational phase of the Point Lepreau monitoring program to distinguish between reactor-derived releases of Cs-137 into the atmosphere and those resulting from even low yield nuclear weapons tests.

Zirconium-95, Niobium-95

Zirconium-95 ($t_{1/2}$ = 65 days) is one of a group of short-lived fission products, generally measurable in the atmosphere for periods of up to a year following a nuclear weapons test. Measurements of the stratospheric inventory of gas particulate Zr-95 and gaseous HTO following high yield thermonuclear (fusion) tests in the 1970's indicated that both radionuclides had about the same half residence time (10 months) in the stratosphere (Mason et al., 1980). These results indicated that gravitational settling of particles in the lower stratosphere is generally unimportant in studies of atmospheric transport. Niobium-95 is the short-lived ($t_{1/2}$ = 35.1 days) daughter product produced by beta decay of Zr-95.

Atmospheric inputs of Zr-95 from the October, 1980 nuclear test first became apparent at the Point Lepreau monitoring sites in January 1981 (Figure 26). Maximized Zr-95 air concentrations occurred during the spring of 1981 with air concentrations decreasing during the summer and falling



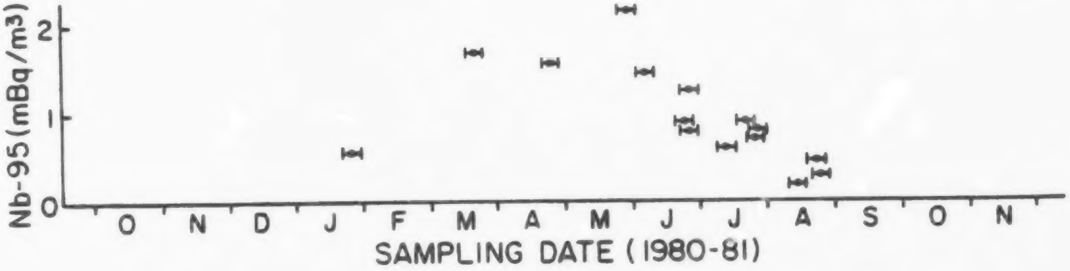


Figure 26. (Upper figure). Zr-95 activities on air particulate material collected at Point Lepreau. Crosses denote EML data. (Lower figure). Nb-95 results for Point Lepreau atmospheric particulates. Summer peaks in both figures due to Chinese weapons test in 1980.

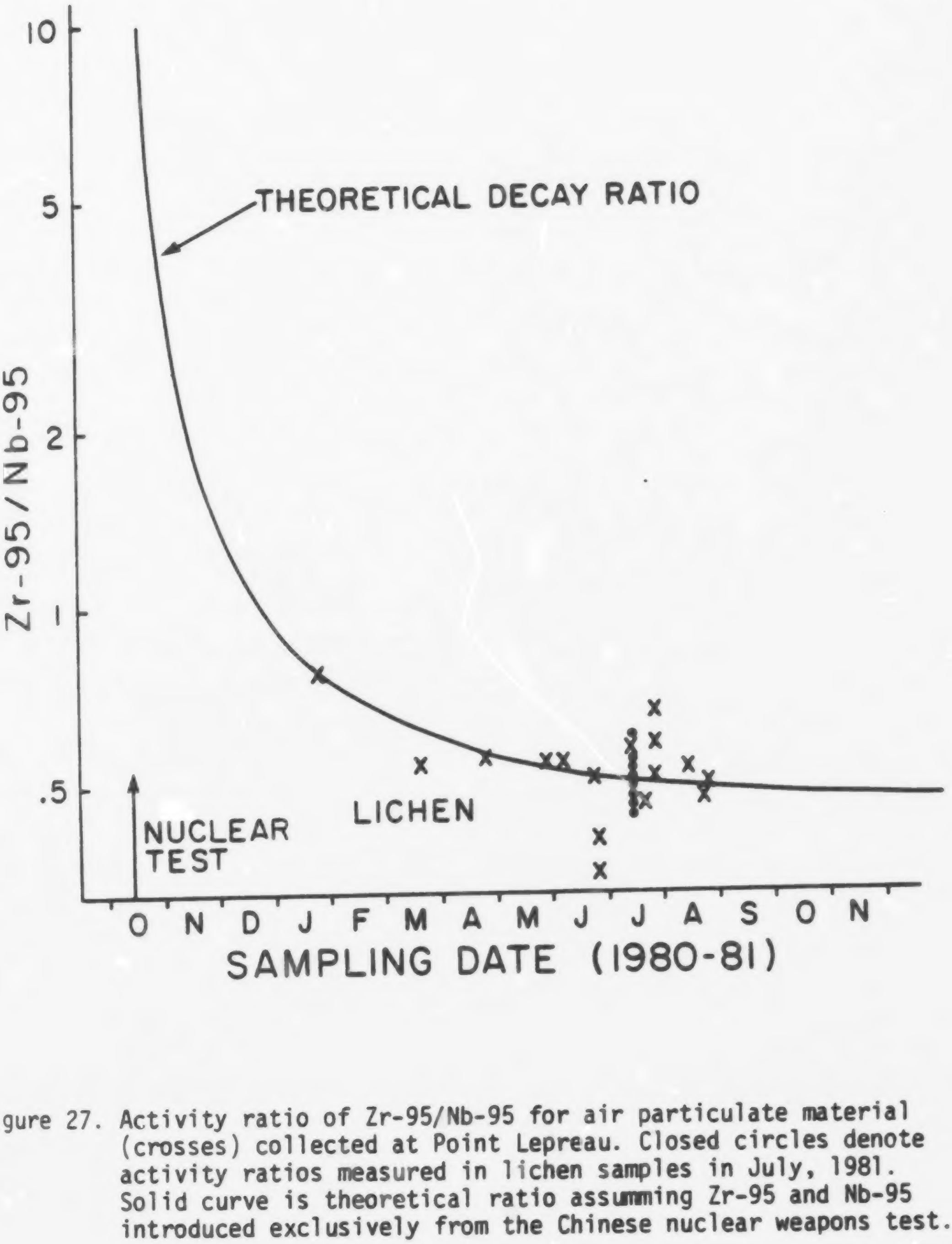


Figure 27. Activity ratio of Zr-95/Nb-95 for air particulate material (crosses) collected at Point Lepreau. Closed circles denote activity ratios measured in lichen samples in July, 1981. Solid curve is theoretical ratio assumming Zr-95 and Nb-95 introduced exclusively from the Chinese nuclear weapons test.

below the detection limit during the fall and winter of 1981. Again, the AERU results for Zr-95 are in reasonable agreement with the EML results for Moosonee, Ontario (Figure 26). The fallout pattern for Nb-95 closely follows that of its parent, Zr-95. Because of its comparatively short half-life, significant ingrowth of Nb-95 can occur between the time of sampling and the analysis of air filters in the laboratory and it must be corrected for. The extent of Nb-95 ingrowth can only be estimated for samples containing a measurable quantity of Zr-95, and for this reason Nb-95 detection limits are not estimated for samples in which the Zr-95 activity is below the detection limit. Estimates of Nb-95 ingrowth from Zr-95 must be performed for all environmental samples analysed more than a few days subsequent to sample collection.

For the general case of ingrowth of a daughter radionuclide (Nb-95) from its parent (Zr-95), the dependence of their activity ratio on time is given by;

$$A_{Zr-95}/A_{Nb-95} = (1 - \lambda_{Zr-95}/\lambda_{Nb-95})\,(1 - e^{-(\lambda_{Nb-95}-\lambda_{Zr-95})t})^{-1} \qquad (1)$$

where λ (= $0.693/t_{1/2}$) is the decay constant for each radionuclide, $t_{1/2}$ corresponds to the half-life, and the initial activity of the daughter (Nb-95) at $t = 0$ is assummed to be neglible. For the case in which the half-life of the daughter is smaller than that of the parent, a condition of "radioactive equilibrium" is eventually attained and equation 1 reduces to;

$$A_{Zr-95}/A_{Nb-95} = 1 - \lambda_{Zr-95}/\lambda_{Nb-95} = 0.46 \qquad (2)$$

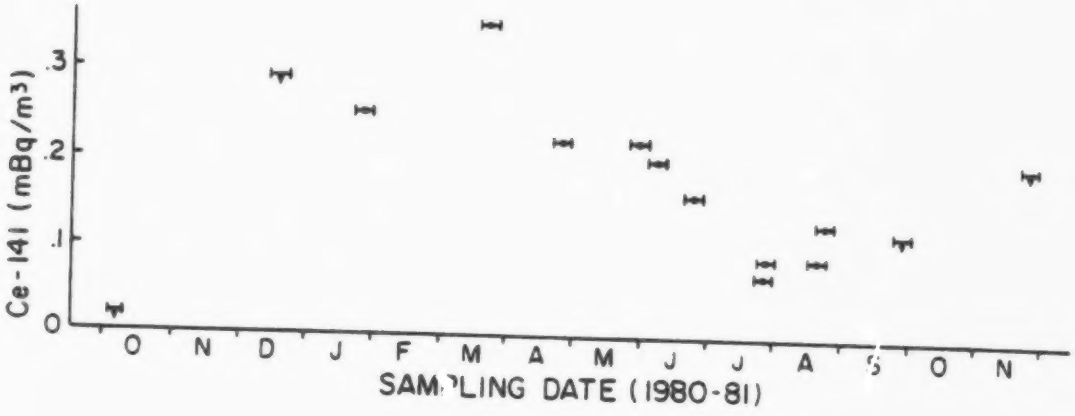
The theoretical dependence of the activity ratio on time for the ingrowth of Nb-95 from Zr-95 produced in the October 16, 1980 weapons test (assuming the initial activity of Nb-95 to be negligible; Cambray et al., 1982) is illustrated in Figure 27. The measured ratios on air particulate material from the AERU air monitoring stations are in excellent agreement with the theoretical predictions between January and August 1981. These results support the contention that there is negligible fractionation between these radionuclides on atmospheric particulate material (either by size fractionation, differential removal from particles by leaching, etc.) and confirm the timing for the initial input of Zr-95 to have been in October 1980. Measurements of the approach of the Zr-95/Nb-95 ratio to radioactive equilibrium are a useful diagnostic for establishing the timing of a fission product input to the environment and this type of analysis may prove useful in identifying environmental inputs of these radionuclides from the Point Lepreau NGS.

Cerium-141, Cerium-144

Cerium-141 ($t_{1/2}$ = 33 days) and Ce-144 ($t_{1/2}$ = 284 days) are fission products which exhibited significant concentration increases in atmospheric particulate material during 1981 as the result of the October 1980 nuclear weapons test. Again, EML measurements of Ce-144 are in good agreement with the AERU results (Figure 28) for the same period of time (Ce-141 was not reported by EML). Since these are isotopes of the same element, negligible fractionation should occur between them in environmental phases and the dependence of their activity ratio on time is given by;

$$A_{Ce-141}/A_{Ce-144} = \overset{*}{A}_{Ce-141}/\overset{*}{A}_{Ce-144} \quad e^{-(\lambda_{Ce-141} - \lambda_{Ce-144})t} \qquad (3)$$

where λ is the decay constant (= $0.693/t_{1/2}$) for each radionuclide and the activity ratio, $\overset{*}{A}_{Ce-141}/\overset{*}{A}_{Ce-144}$, corresponds to the relative activities of the two isotopes at $t = 0$, the time of their initial introduction into



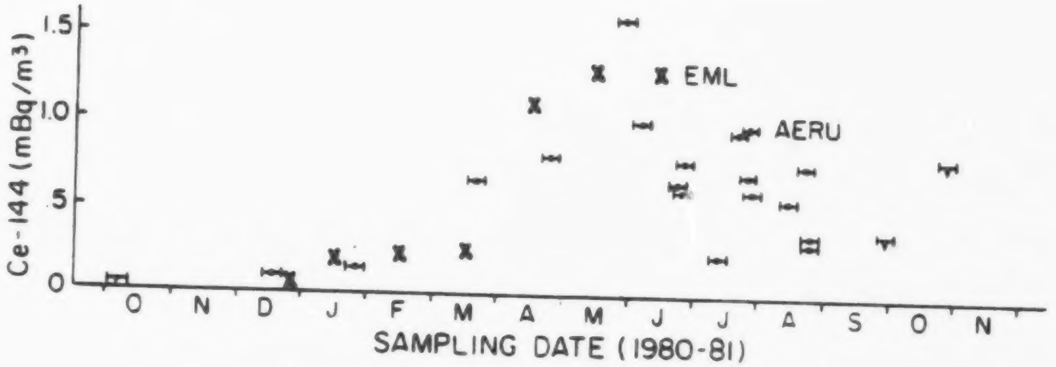


Figure 28. (Upper figure). Ce-141 activities on air particulate material. Width of bars denotes sampling interval; arrows denote detection limits. (Lower figure). Ce-144 activities on air particulate material. Crosses correspond to EML results (Moosonee, Ont.).

the environment. The fission product ratios resulting from nuclear weapons tests will vary, depending on the nature of the weapons device, but in the past an initial ratio of $A^{*}_{Ce-141}/A^{*}_{Ce-144} = 8.4$ has proven to be a reasonable estimate for a Pu-239 trigger explosion (Harley et al., 1965). Using this value for the initial ratio, the theoretical dependence of the fission product ratio on time is compared in Figure 29 to the ratios measured for atmospheric particulates during 1981. The measured values, uniformly higher than this ratio, may be due to an initial fission yield ratio for the October 1980 weapons test of slightly greater than 8.4. The measured rate of decrease in the isotope activity ratio with time is close to the predicted rate, although the ratios measured in June-August are higher by a factor of two than their predicted values. However, the general agreement between the predicted and measured activity ratios is consistent with an October, 1980 input of fission products and also indicates that background levels of the longer-lived Ce-144 on atmospheric particulate material are low. This latter conclusion follows from the fact that a significant background inventory of Ce-144 on atmospheric particulates derived from previous weapons tests would result in lower measured Ce-141/Ce-144 ratios compared to predicted ratios. It is interesting to note that the Ce-141/Ce-144 ratios measured in lichen samples are lower than the ratios measured on air particulate material collected at approximately the same time. This is probably due to enhanced background levels of Ce-144 accumulated by the lichen from previous weapons tests as is discussed in a following section of this report.

Ruthenium-103, Ruthenium-106

Ruthenium-103 ($t_{1/2} = 39.4$ days) from the October 16, 1980 weapons test was first detected in atmospheric particulate material in January

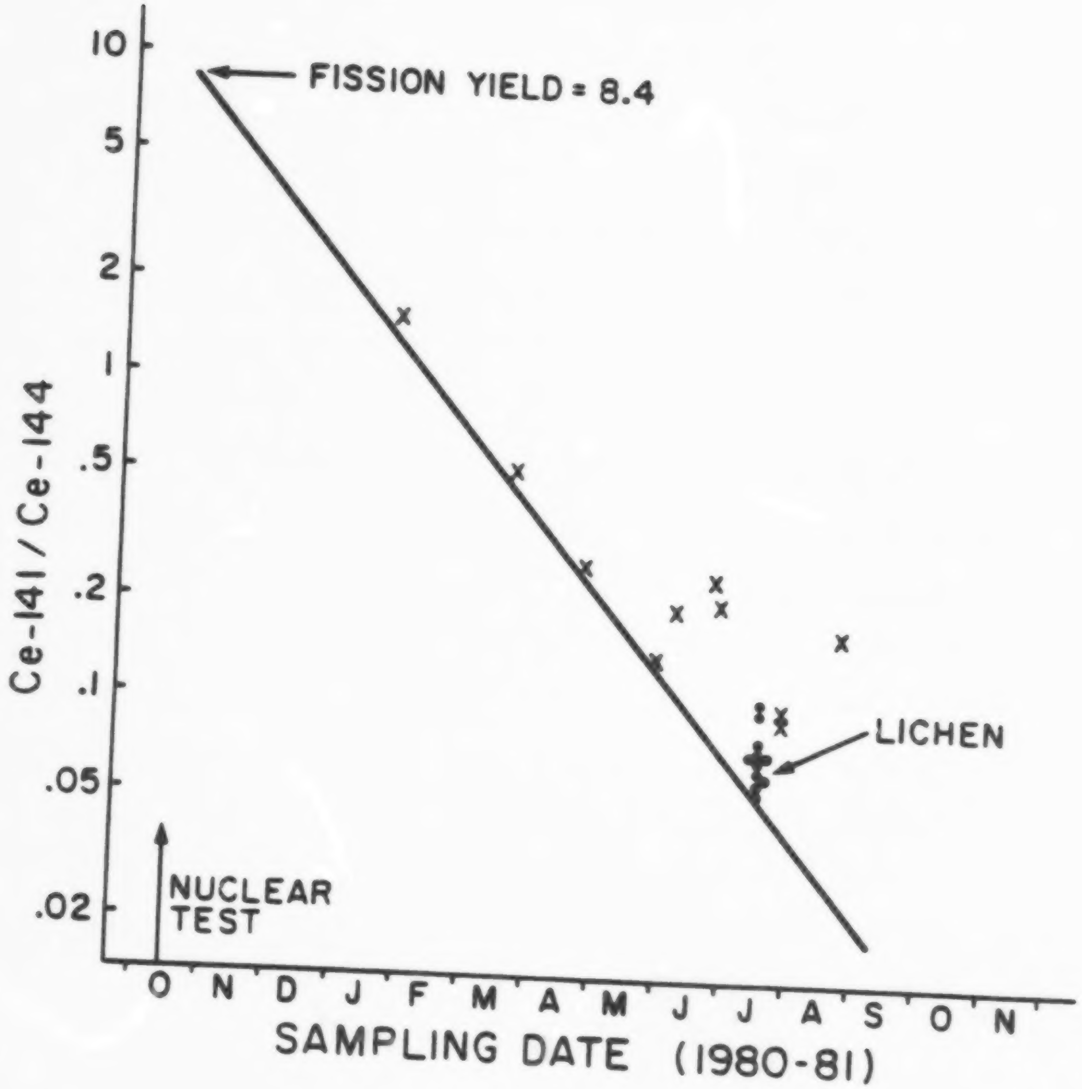
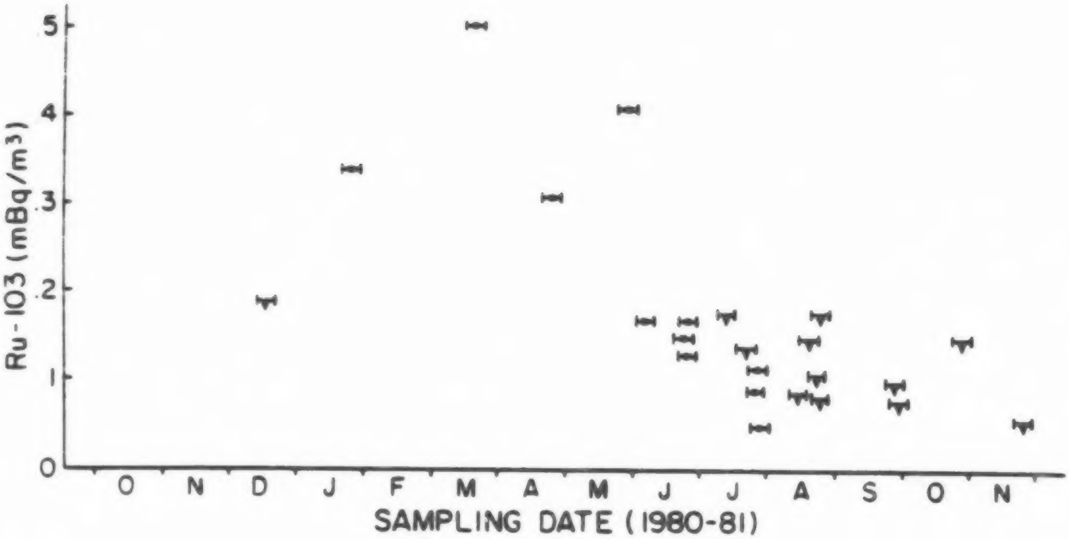


Figure 29. Isotopic ratio of Ce-141/Ce-144 for air particulate material. Theoretical decay ratio for Chinese nuclear test (Oct., 1980) given by solid curve assumming initial fission yield of 8.4 for this isotopic ratio. Closed circles denote isotope ratios in lichen samples collected near Point Lepreau.

1981 (Figure 30) and the highest concentration was measured in March 1981. The longer-lived isotope, Ru-106 ($t_{1/2}$ = 368 days) was first detected at Point Lepreau in May 1981 with air concentrations of both radionuclides falling below the detection limit by August 1981. Although there is only a limited quantity of Ru-106 data, it is interesting to examine the time-dependence of the Ru-103/Ru-106 activity ratio. The initial fission activity ratio for these isotopes as the result of the 1980 weapons test is 19.7 (Harley et al., 1965). As in the case of the cerium isotopes, the activity ratio, A_{Ru-103}/A_{Ru-106} should decrease exponentially as a function of time, according to equation 3. The measured Ru-103/Ru-106 ratios measured in air particulate material during the summer of 1981 are in excellent agreement with ratios predicted on the basis of the 1980 weapons test (Figure 31). Again, this good agreement unambiguously identifies the source of the atmospheric particulate inventory of radioactive ruthenium to be the Chinese explosion. As in previous cases, the ruthenium isotopic ratio in lichen material is reduced compared to air particulate values, indicating that there is a residual background of the longer-lived isotope, Ru-106, on lichen derived from previous weapons tests.

Fission Product Ratios

Fission product activity ratios for different elements following the nuclear weapons test in 1980 should also vary according to equation 3, although in this case there may be some degree of fractionation between these elements in different environmental phases. The extent to which the different radioactive elements follow separate environmental pathways will be reflected in the divergence of their activity ratios from the predicted values.



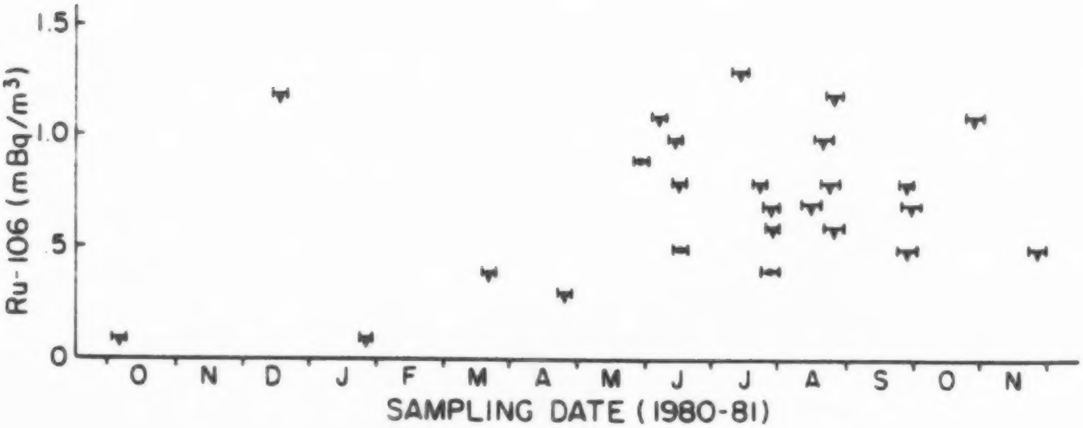
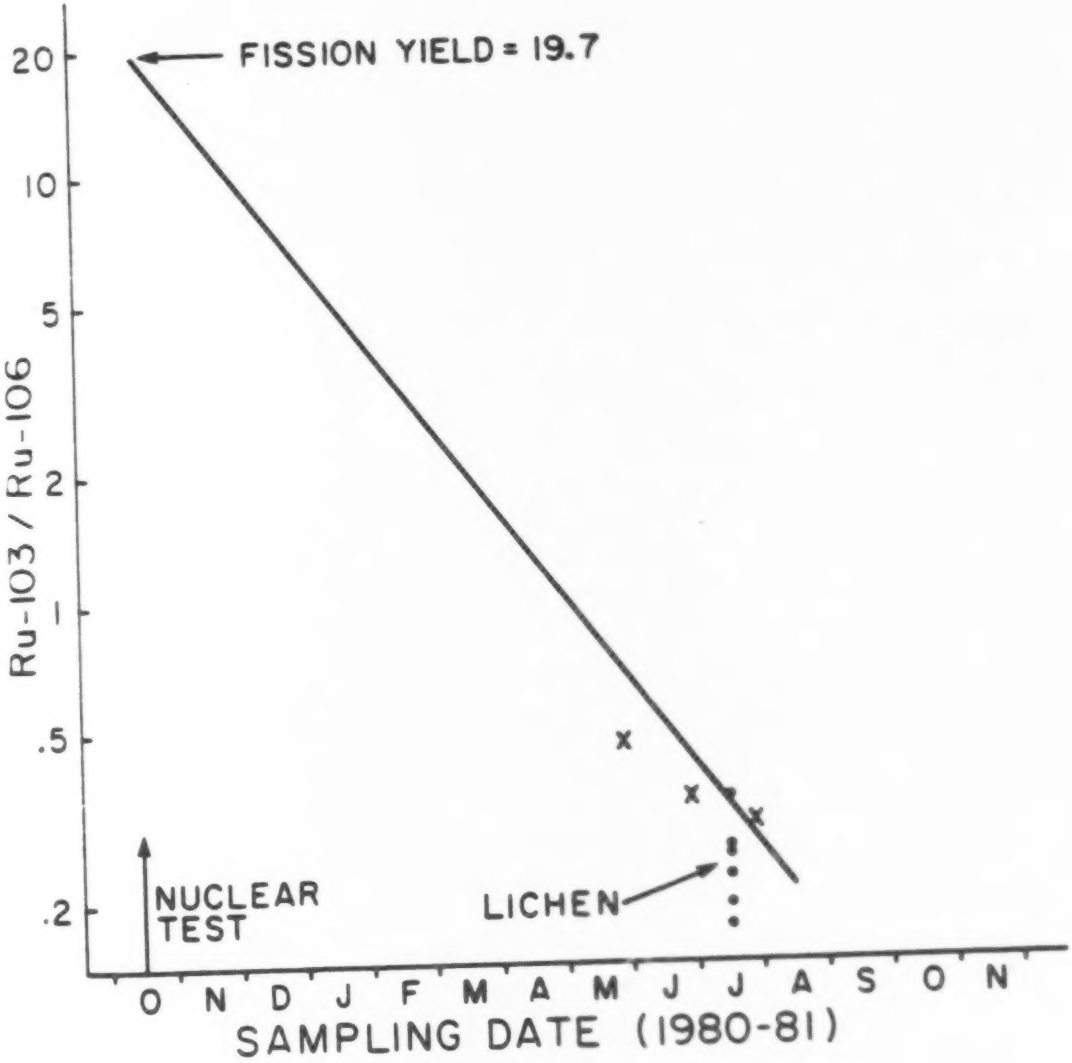


Figure 30. (Upper figure). Ru-103 activities for air particulate material. Maximum measured during spring, 1981, caused by Chinese nuclear weapons test in 1980. (Lower figure). Ru-106 activities for air particulate material with most values being below detection limit.



gure 31. Crosses denote measured Ru-103/Ru-106 activity ratios for air particulate material. Circles denote activity ratios for lichen samples. Solid curve is theoretical decay ratio as function of time.

The Ru-103/Zr-95 ratio on air particles (Figure 32) decays according to the respective decay constants of the two isotopes and is in excellent agreement with the predicted curve based upon the initial fission yield of 1.68 given by Harley et al. (1965). These results indicate that there is negligible fractionation between these two elements on air particulate material. Further, the activity ratios measured on lichen in July, 1981 are in reasonable agreement with ratios measured on air particles at approximately the same time. This result indicates that the lichen are good samplers for atmospheric particulates and that little differential removal of either Ru or Zr occurs as a result of leaching subsequent to their deposition on lichen surfaces.

The decrease in the Ru-103/Ce-144 ratio with time (Figure 33) is also in good agreement with the predicted decrease following the 1980 weapons test. However, the ratio of these isotopes in lichen samples is considerably lower than the value for the same time period for atmospheric particulates. Although this could be due to fractionation between these elements subsequent to their deposition, it is more likely that the lower lichen values are due to residual amounts of the longer lived Ce-144 from nuclear weapons tests carried out prior to the 1980 test. It follows from the above comparisons that the activity ratio, Zr-95/Ce-144 (Figure 34) is also in good agreement with the theoretical decay curve. Again, the lower values for this ratio measured in lichen compared to those measured in atmospheric particulate material are probably due to background (pre-1980) inputs of Ce-144.

The initial fission yields for Zr-95/Cs-137 (156) and Ce-144/Cs-137 (32.9) are again based upon estimates made by Harley et al. (1965). The ratios measured on atmospheric particulates were lower than the

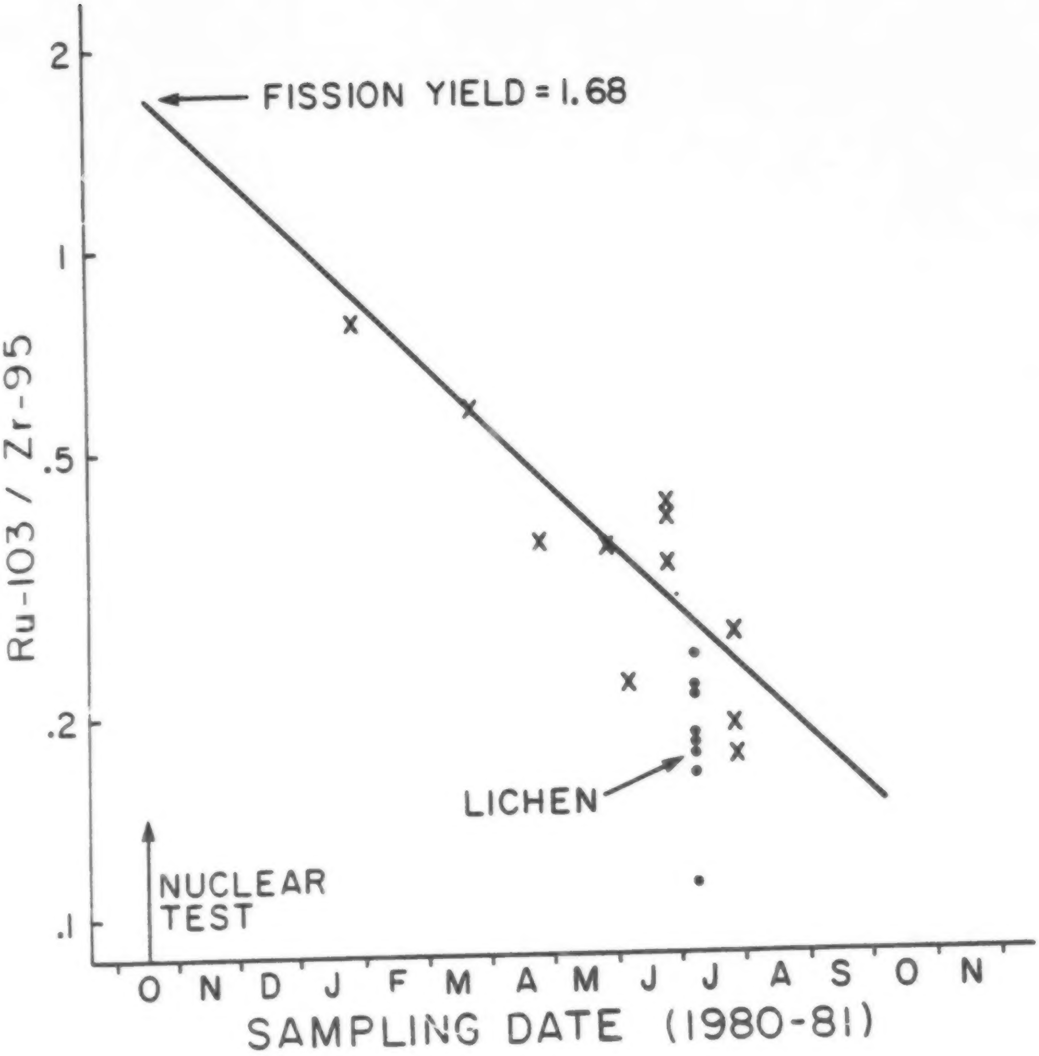


Figure 32. Crosses denote measured Ru-103/Zr-95 ratios for air particulate material. Closed circles denote ratios measured for lichen samples. Solid curve is theoretical decay ratio for material introduced during 1980, Chinese nuclear weapons test.

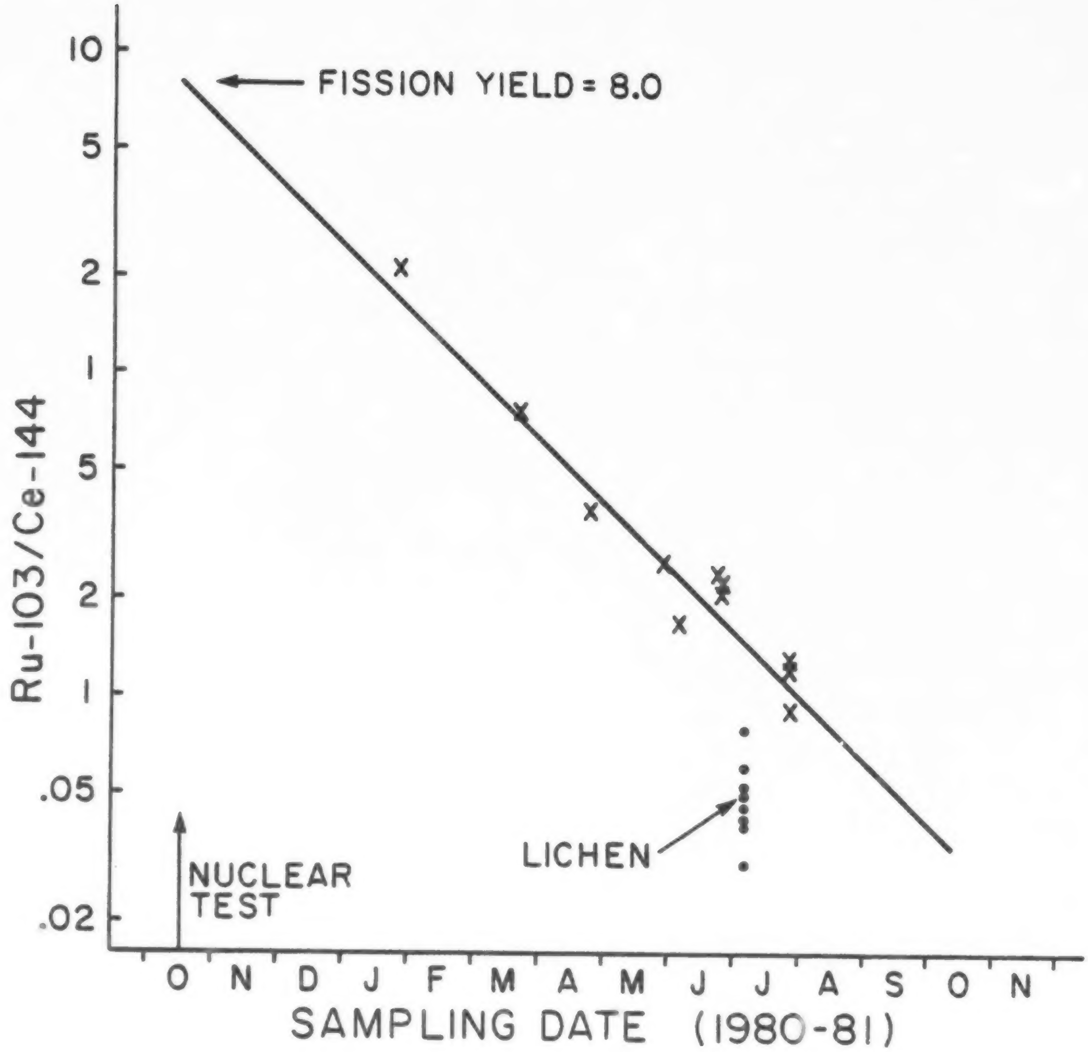
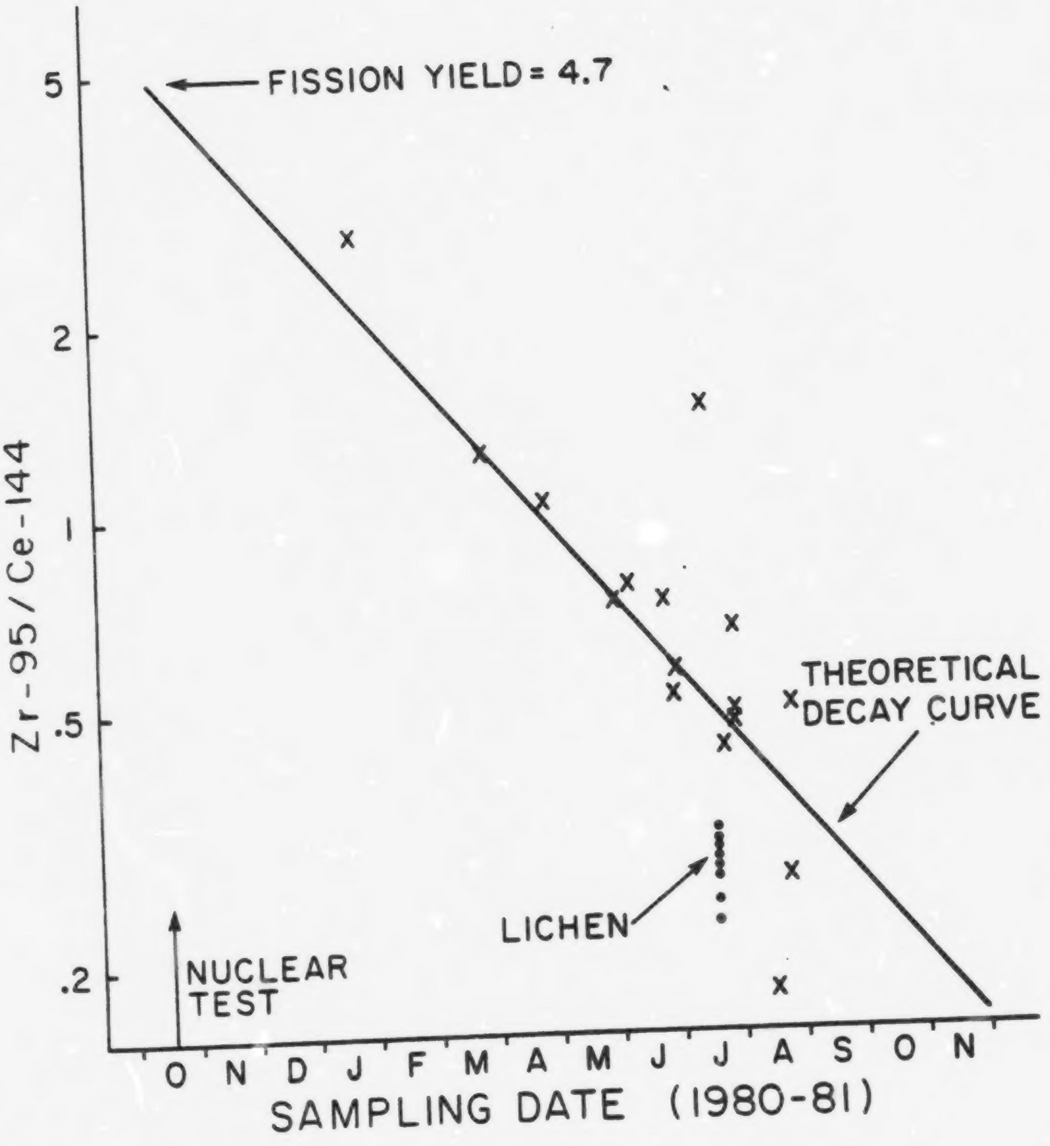


Figure 33. Crosses denote measured Ru-103/Ce-144 activity ratios for air particulate material. Closed circles denote activity ratios for lichen samples. Solid curve is theoretical decay ratio for 1980 nuclear weapons test. Lowered lichen activity ratios denote residual background of Ce-144 from previous weapons tests.



igure 34. Crosses denote measured Zr-95/Ce-144 activity ratios for air particulate material. Closed circles denote activity ratios for lichen samples. Solid curve is theoretical decay ratio for nuclear weapons test assumming initial activity ratio of 4.7 in October, 1980.

predicted values during the summer of 1981 by a factor of the order of 30% (Figures 35 and 36). These results indicate that approximately 30% of the Cs-137 inventory on air particulate material was a residual background component derived from previous nuclear tests, a result which would be expected in view of the comparatively long half-life of Cs-137. The ratios of these radionuclides in lichen samples are considerably lower than the ratios measured on air particles. This is to be expected since the long life time of the lichen (5-20 years) means that much of the Cs-137 sorbed onto the lichen surface is derived from nuclear weapons tests carried out in the 1960s. Hence, ratios of short lived fission products compared to Cs-137 on lichen should be an order of magnitude smaller than activity ratios on air particulates.

The good agreement between predicted and measured activity ratios for these fallout radionuclides indicates that the air monitoring stations are functioning effectively and that the air particulate samples are representative of the atmospheric loading. Further, these results illustrate the usefulness of isotope ratios as a diagnostic to determine the source and the timing of radionuclide inputs into the environment. By following the changes in isotope ratios as these elements pass through different environmental phases, it is possible to identify different transport pathways and to determine accumulation rates of these elements in various environmental matrices.

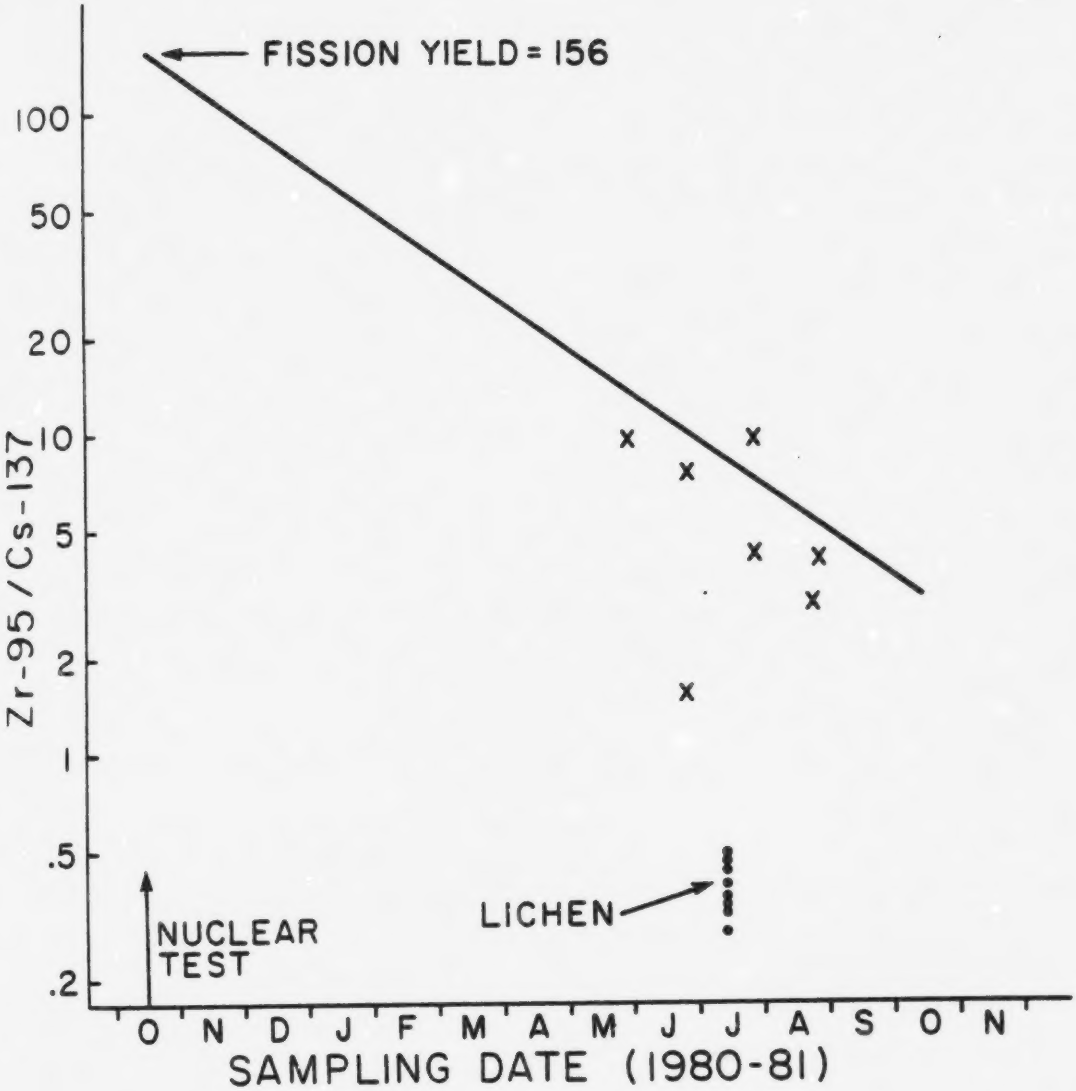


Figure 35. Crosses denote measured Zr-95/Cs-137 activity ratios for air particulate material. Activity ratios for lichen samples (closed circles) reduced due to presence of background Cs-137 in lichen from previous weapons tests. Solid curve represents theoretical isotopic decay ratio for Oct., 1980 nuclear test.

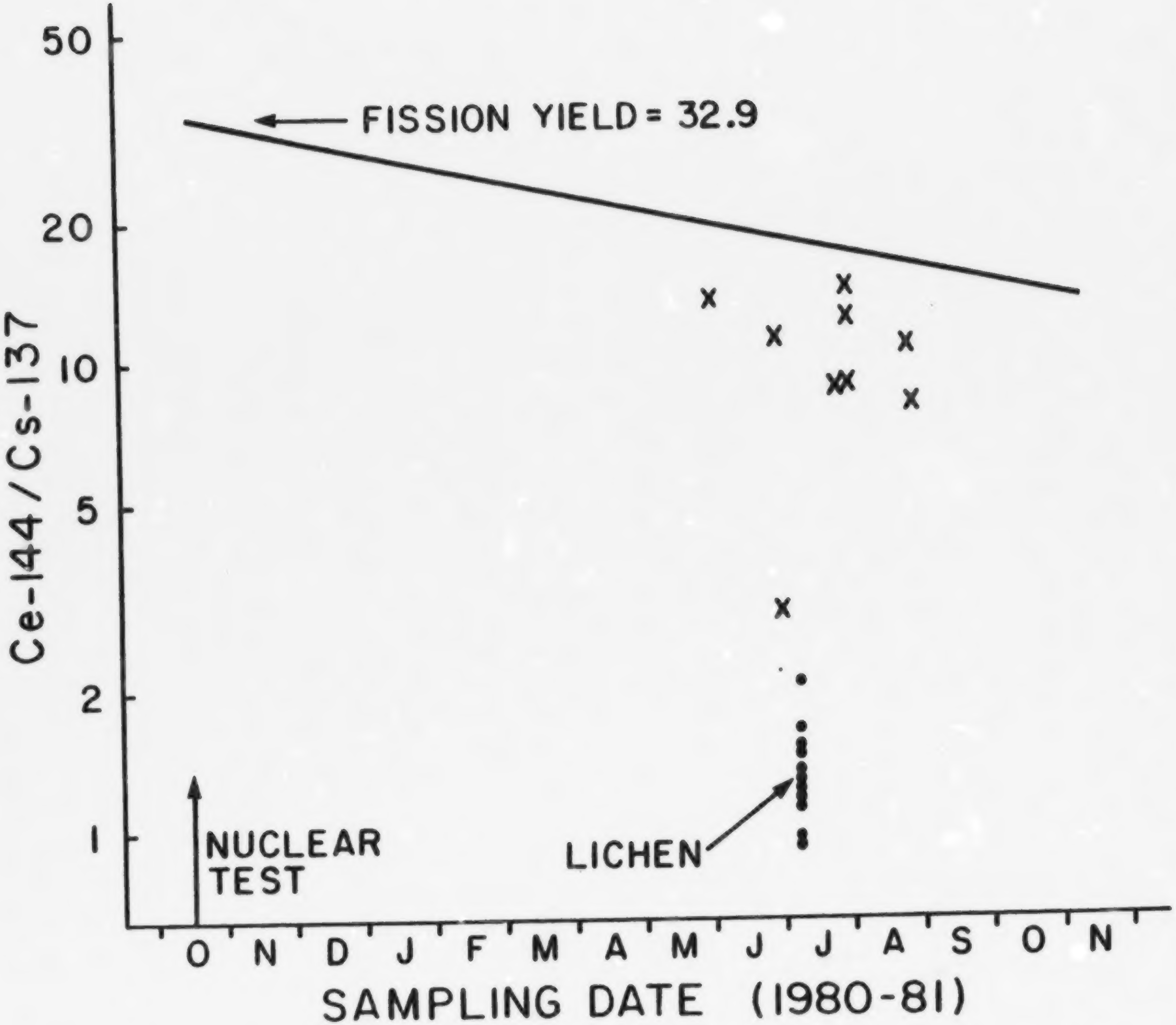


Figure 36. Crosses denote measured Ce-144/Cs-137 activity ratios for air particulate material. Activity ratios for both air particles and lichen samples (closed circles) are smaller than theoretical decay ratios (solid curve) due to residual background level of

5.0 BIOLOGICAL OCEANOGRAPHY

One of the principal concerns associated with the operation of the Point Lepreau NGS is the effect produced on biological phases of the environment by the release of heated effluent and radioactivity. The thermal plume will be largely confined to the immediate vicinity of the reactor outfall where sublethal effects on organisms may include changes in biological functions such as growth, respiration rate, reproduction and behaviour. These changes could manifest themselves in changes in species diversity and community structure near the outfall. Since benthic organisms are relatively immobile and form characteristic assemblages related to specific water quality parameters, these organisms are useful as indicators of ecological effects. Consequently, several studies have been carried out (described in Smith et al., 1981) to provide information on potential future perturbations in ecological parameters associated with benthic communities at Point Lepreau as the result of discharges from the reactor. One study involved in assessment of the effects produced on benthic communities since 1974 by the release of heated water from the Coleson Cove Thermal Generating Station (located 15 km east of Point Lepreau on the Bay of Fundy coast) and the second was an ecological baseline study of benthic communities near the Point Lepreau NGS outfall. Similar studies will be performed subsequent to the reactor start-up in order to identify any deleterious ecological effects caused by the reactor thermal plume.

5.1 Biological Uptake of Radioactivity

Most of the radionuclides emitted from nuclear reactors are non-conservative elements (with the important exception of tritium) which do not necessarily follow water circulation patterns and whose transport processes are influenced by plants and animals. Biological uptake of nonconservative elements is generally greater than that for conservative elements, the latter being elements which are present in seawater in amounts directly related to the salinity of the water and whose transport and distribution in the oceans is largely controlled by physical processes. Marine organisms may concentrate some elements to levels many times those in seawater and these organisms may be capable of moving independently of water circulation patterns and redistribute their net accumulation of trace elements and radioactivity throughout different levels of the food chain. The term "concentration factor" has been used to characterize biological uptake of chemical species and may be defined as the ratio of the concentration of an element or radionuclide in an organism (or its tissues) to that concentration directly available from the organism's environment under steady-state conditions. This definition of a concentration factor is complicated by the fact that aquatic organisms do not normally derive all of their nutrients or radionuclides from one source, but instead accumulate them from a variety of sources including food, water and suspended or deposited sediment.

For many organisms, the concentration factors reported in the literature are based upon the preceeding food link. However, in the present report, the concentration factor (C.F.) refers to the radionuclide concentration in the tissue of an organism divided by its concentration in seawater,

$$\text{C.F.} = (100 - \%\ H_2O)\ A_{{Cs-137}_{sample}} / 100\ A_{{Cs-137}_{water}} \qquad (4)$$

where, $\%H_2O$ is the % wet weight, $A_{{Cs-137}_{sample}}$ is the Cs-137 activity of the sample expressed as the sample dry weight and $A_{{Cs-137}_{water}}$ is the mean Cs-137 activity measured in solution in the lower Bay of Fundy. This is a valid concept for phytoplankton which accumulate nutrient elements directly from water, in which case their radionuclide content may be directly compared with that of seawater for the purpose of concentration factor estimates without the complication of intervening trophic levels. For organisms at higher trophic levels the radionuclides may be preconcentrated or discriminated against as they pass through different trophic levels before being incorporated into the individuals of a given link in the food web. However, even for species constituting higher levels of the trophic levels, a steady-state may be established between the animals, their food and the radionuclide concentration in the water, in which case concentration factors based on sea water values are useful indicators.

Concentration factors are less useful for the case of infaunal marine organisms that live in sediments and feed on detritus. Many of these organisms ingest large amounts of sediment along with organic detritus and one is faced with the problem of selecting the environmental basis against which to compare the radionuclide concentration in the organism. Further uncertainty is introduced by the fact that the concentrations of many dissolved elements in the interstitial waters of sediments differ significantly from those in the overlying seawater. Despite these drawbacks, concentration factors provide a semi-quantitative measurement of the accumulation of radionuclides in different organisms and are useful as indicators of radionuclide releases.

The time required for a steady-state distribution of a radionuclide between an organism and its surrounding environment to be established (a distribution characterized by the concentration factor) is directly related to the biological turnover rate of the radionuclide or its stable element counterpart and inversely proportional to the rate of environmental dilution. The length of time required for an organism to exchange half of its total content of a given element is referred to as the biological half-life of the element. The shorter the biological half-life, the faster the organism may achieve a steady-state type of equilibrium with respect to seawater radionuclide concentrations. For the case in which radioactivity levels in the water column are variable as a function of time (depending, for example, on the release rates of radionuclides from a nuclear reactor) radionuclide levels in organisms living in the area will follow changes in seawater radionuclide levels at different rates, each dependent on a particular turnover rate. Organisms with comparatively long turnover times (such as animals in higher trophic levels) may be unable to achieve steady-state with environmental levels of radioactivitv. Smaller marine organisms, such as plankton, frequently have turnover rates for trace elements with half-times measured in hours and for these organisms, "equilibration" with changing levels of a radionuclide in the water may be achieved.

A third consideration in understanding the degree to which a radionuclide is concentrated by an organism is the extent to which physiochemical interactions between the element and the organism itself control the biological uptake process. Goldberg (1957) showed that the ability of marine organisms to concentrate metals from the sea paralleled the order of metal ligand complexes, and Szabo (1967) reported that mixed plankton accumulated alkaline earths in the same sequence as cation exchangers such as Dowex-50 in which the order of association is radium > barium > strontium > calcium > magnesium. It is well known that alkali metals (potassium, sodium, rubidium and cesium) may be incorporated into bottom sediments as the result of ion-exchange reactions with clay minerals and it has been proposed that similar ion-exchange reactions may occur at the surfaces of marine organisms. Surface adsorption by ion exchange may be particularly important for the uptake of alkaline earths and alkali metals by phytoplankton and additional elements may be adsorbed by chelating processes.

5.2 Processing and Analysis of Biological Materials

An assessment of environmental effects associated with the biological uptake of radioactivity requires a surveillance program consistent with local and regional marine energetics, the nature of the food web, water circulation and zones of productivity. Sampling sites have been chosen for the collection of representative species as outlined in Table 10. Biological samples have been collected from the major marine habitats in the Point Lepreau region and these samples have been analysed for radioactivity in the AERU laboratory. Sampling methods, criteria and frequency have been noted in Bishop et al. (1980), although these procedures have been modified to some extent as described below. In general, species have been selected for surveillance which occur abundantly in most of the major habitats and which play a significant role in the food web. All trophic levels are represented and any trends in the transport and accumulation of radioactivity within these communities should become apparent at some level within the sampling assemblage.

Marine biological samples were frozen as soon as possible subsequent to collection and transported in this state to the AERU laboratory. Plant samples were freeze-dried immediately while animal samples were allowed to thaw and specific components such as shell, muscle, liver, brain

Table 10. Representative species for different marine sampling regimes near Point Lepreau.

SPECIES	EMBAYMENT, ESTUARY	INTERTIDAL ZONE	SUBTIDAL ZONE	PELAGIC ZONE
Plants		Rockweed, irish moss dulse, Fucus	Laminaria	
Grazers	Periwinkle			
Detritus Feeders		Gammarus		
Suspension Feeders	Soft shelled clam	Blue mussel Hermit crab	Horse mussel	
Predators and Scavengers	Sandshrimp	Green crab	Rock crab	Rock crab
Fish		Flounder		Herring
Birds	Purple sandpiper	Eider duck	Cormorant, Herring Gull	
Mammels			Harbour seal	
Planktonic Forms				Bulk Samples Phyto- and Zooplankton

or hepatopancreas were separated from the tissues. These parts were homogenized (using a food processor or commercial blender) and in some cases, 50 g of material removed for tritium analyses. Soft tissues were freeze-dried while shell and bone material was dried in an oven with the percent moisture content determined for all cases. Subsequent to processing all samples were kept in air tight plastic containers.

In order to determine the tritium content of marine biological materials, soils, and terrestrial plants and organisms it was necessary to extract the water from these samples under non-contaminating conditions. To meet this requirement an azeotropic distillation procedure (Moghissi et al., 1973) was followed in the AERU laboratory. Briefly, the method involves the addition of scintillation grade toluene to a solid (or liquid) sample in a distillation flask. With the application of heat, the distillation proceeds at 85° C and the vapour, consisting of 20% water and 80% toluene, is condensed and collected in a vessel where the two components separate. Completion of the extraction is indicated by a rise in the temperature of the vapour to the boiling point of toluene (110° C). The extracted water is then analysed for tritium following the standard liquid scintillation procedure. The remaining bulk sample may be analysed for other radionuclides after removal of the toluene by distillation and heating in a steam bath. Tritium analyses performed on samples collected prior to 1981 revealed that the biological materials had been contaminated with tritium during handling and storage. Additional samples will be collected prior to the operation of the reactor and analysed using the method outlined above to establish biological baseline levels of tritium.

Samples were analysed for gamma emitting radionuclides using a 21% efficient germanium-lithium detector. A gamma spectra for each sample was automatically analysed using a PDP-11-04 computer which identified and quantified radionuclide peaks of interest. Samples were analysed using one of five different standard geometries, (the choice of which depended on the available sample size) corresponding to sample volumes of 500 ml, 300 ml, 100 ml 50 ml, and 20 ml. The effect of self absorption on counting efficiency due to different sample density was found to be negligible for the range encountered in the biological samples. Detection limits (Bq/Kg) for each radionuclide of interest are given in Appendices 2 and 3 for the five standard counting geometries employed in these analyses. These detection limits correspond to a counting time of 100,000 sec, for analyses performed one month subsequent to the sampling date (Appendix 3) and one year subsequent to the sampling date (Appendix 4). Note that for the longer time period between sample collection and analysis, many of the shorter-lived radionuclides are virtually undetectable owing to their radioactive decay. Appendices 3 and 4 can be used to calculate radionuclide detection limits for any sample listed in the following sections of this report. The standard counting geometry for each sample is used to identify the general detection limit for the isotope of interest from Appendices 3 and 4. This general detection limit is then divided by the sample weight to determine the specific detection limit for the sample. These detection limits are for typical counting conditions and will vary slightly with changes in the above parameters. The use of these tables as outlined above simplifies reporting procedures because it eliminates the necessity of reporting each radionuclide detection limit for all samples analysed in the AERU laboratory.

5.3 Radionuclide Results - Plankton

Partly because of their large surface area, plankton have large concentration factors for many elements. Further, their rapid growth rates lead to fast response times for radionuclide accumulation which make plankton excellent indicator species for changes in radioactivity levels in seawater. Plankton also occupy a critical position at the base of the food chain and their uptake of radionuclides will invariably result in increased exposure to organisms in higher trophic levels. Phytoplankton are primary producers which principally take up radionuclides by adsorption, although a large portion of trace elements are accumulated by absorption processes. Zooplankton accumulate radionuclides by both processes in addition to uptake through consumption of phytoplankton. In areas contaminated by radioactivity in the Adriatic Sea (Jelisaveic, 1970) and near Cap de la Hague, France (Germain et al., 1979) plankton have been shown to be particularly effective in concentrating Ce-144, Ce-141, Ru-106 and Zr-95 at levels up to 3500 Bq/kg. Cesium-137 and Co-60 were measured at levels of up to 200 Bq/kg and trace levels of Zn-65, Sb-125, Fe-59 and Mn-54 were observed. Zooplankton radioactivity levels are generally lower than those in phytoplankton owing to their relatively inefficient assimilation of radionuclides across the gut wall.

Plankton populations in the vicinity of Point Lepreau are unusually low for coastal waters (Gran and Braarud, 1935; MacLaren Atlantic Ltd., 1977). The high suspended particulate matter load in the Bay of Fundy and the large quantity of fog which reduces the amount of light available to phytoplankton are responsible for their low populations. Zooplankton populations are low because of the decreased phytoplankton populations and because of the sensitivity of their egg and larvae to low temperature (Pentreath, 1980). The lowest plankton populations were recorded in the fall, winter and spring seasons (average phytoplankton population of 1400 cells/l; zooplankton of 130 cells/l) and the highest in the summer (average phytoplankton of 22,000 cells/l; zooplankton of 700 cells/l). Generally, the phytoplankton population is dominated by diatoms and dinoflagellates with only minor representation by other groups. The zooplankton population is dominated by several species of copepods with occasional inclusion of other groups.

Plankton samples were collected during plankton tows along transects indicated in Figure 37 using plankton nets which were of 64 μm mesh size for phytoplankton and 333 μm mesh size for zooplankton. No attempt was made to distinguish between the detritus and suspended particulate material and the plankton recovered in the nets or to determine the dominant plankton species in each sample. Several difficulties were encountered in sample collection and preparation for analysis. First, the low population levels of plankton in the Bay of Fundy required plankton tows of many hours duration in order to obtain samples of adequate weight for radionuclide analysis. Secondly, filtering of the samples occasionally causes rupture of the cell walls and the subsequent release of internal radioactivity, especially after freezing. Hence, plankton samples collected in 1980 were freeze-dried in the seawater in which they were collected and the final dry sample contained large quantities of salt (up to 75% of the sample by volume). This problem was reduced for post-1980 samples by carefully screening the plankton sample with a net in order to remove the salt water immediately after sample collection.

Results for gamma-emitting radionuclides measured in plankton samples are given in Table 11. Water content in the samples was high

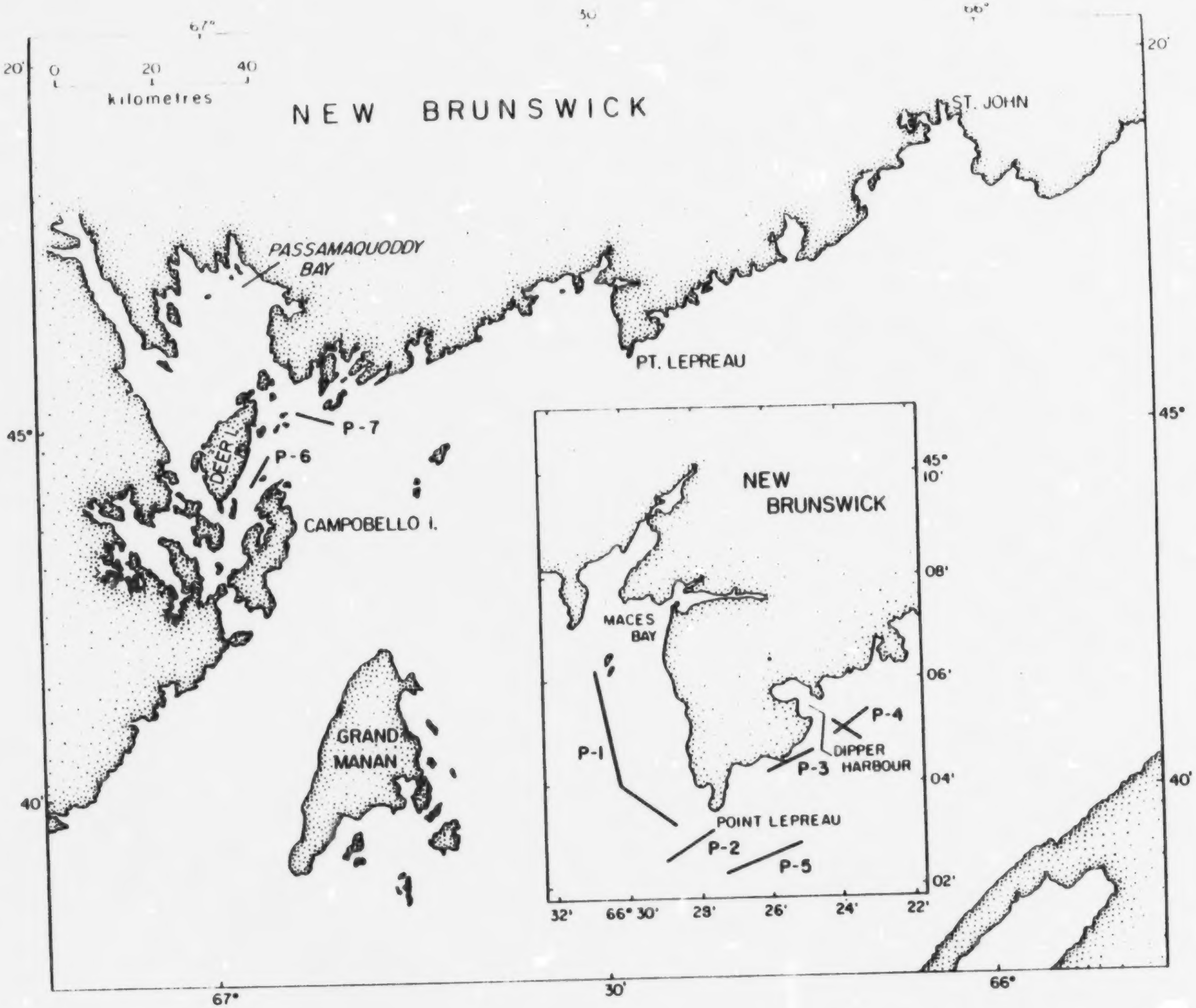


Figure 37. Transects corresponding to sample tows for plankton carried out in western Bay of Fundy.

Table 11. Radionuclide results for plankton samples. Plankton were not separated from detritus and SPM and the two categories are distinguished solely by the plankton net mesh size.

Organism	Sample No.	Organ	Collection		Sample		%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
			Site	Date	Weight (g)	Geometry				
600 µm mesh plankton net sample ("zooplankton")	80-214	total	P-1	6/2/80	25.76	3	95.8	<2.7	<34	<3.3
	80-215	total	P-2	5/2/80						
	80-216	total	P-3	5/2/80						
	80-217	total	P-4	8/2/80						
	80-218	total	P-4	8/2/80						
	80-219	total	P-4	8/2/80						
	80-316	total	P-6	24/10/80	72.95	1	97.0	<1.0	<9.0	<1.2
	80-317	total	P-7	24/10/80						
	80-318	total	P-3	3/11/80						
	82-8655	total	P-5	14/5/82	1.67	5	97	<12	<121	<15
64 µm mesh plankton net sample ("phytoplankton")	80-220	total	P-1	6/2/80	18.57	3	96.5	2.3 ± 1.8	<23	<8.0
	80-221	total	P-2	5/2/80						
	80-222	total	P-3	5/2/80						
	80-223	total	P-4	8/2/80	22.05	3	97	<2.0	<18	<3.8
	80-224	total	P-4	8/2/80						
	80-225	total	P-4	8/2/80						
	80-320	total	P-6	24/10/80	29.40	3	96.0	<2.0	<24	<3.5
	80-321	total	P-7	24/10/80						
	80-322	total	P-3	3/11/80						
	82-8651	total	P-5	13/4/82	20.08	3	97	<3.1	<31	<3.3
	82-8652	total	P-5	13/4/82	9.05	3	97	<7.2	<72	<7.3

(96%-97%) and artificially-produced radioactivity was measured in only one sample. The concentration factor of Cs-137 in plankton estimated from the one measurement above background levels is in reasonable agreement with literature values.

5.4 Marine Algae

Marine algae are primary producers located at the base of a large food web and have high concentration factors for many non-conservative elements, factors which make them commonly used indicators of marine radioactivity and trace elements. They also have several practical advantages compared to other indicator species because large sample sizes are easily collected and their absense of mobility makes them ideal as site-specific indicators for reactor effluents. Germaine et al. (1979) studied radionuclide uptake in a wide range of marine organisms near the Cap de la Hague nuclear fuel reprocessing plant in France and measured the highest concentration factors in marine algae. Uptake of Ru-106 released from the Windscale Reprocessing Plant by *Porphyra umbilicales*, a marine algae used to make laverbread which is consumed by a large population in Great Britain, constitutes a classic case of the "critical pathway" for radioactivity exposure to the public from this facility. Uptake of radioactivity by algae is accomplished by adsorption onto the plants surface where it is firmly bound. Species commonly used for monitoring include *Porphyra umbilicales*, *Ascophyllum nodosum*, *Fucus veseculosis*, *Fucus serratus*, *Ulva lactuca* and *Laminaria sp.* The degree of accumulation of radionuclides varies from species to species, but most are suitable as indicators. Marine algae are particularly useful in monitoring I-131 because of its high concentration factor (up to 20,000) and because I-131 is not easily measured in other phases of the environment.

Activity ratios of isotopes of the same element can be used to determine integration factors for different algae collected at the same location. Aarkrog (1979) estimated integration times (similar to a biological turnover time) using Co-60/Co-58 ratios for algae and found the order of integration times to be *Fucus vesiculosis* = *Fucus spiralis* < *Fucus serratus* < *Ascophyllum nodosum*. The same estimates may be performed using Cs-137/Cs-134 and Ce-144/Ce-141 ratios.

Marine plants selected for monitoring at the Point Lepreau NGS include the brown algae, *Ascophyllum nodosum*, *Fucus vesiculosis*, *Laminaria* and the red algae; *Chondrus crispus* and *Rhodymenia palmata*. *Ascophyllum nodosum* is the most abundant algae on the rocky shoreline near Point Lepreau where it is found in the mid-littoral zone and provides food primarily to grazers such as periwinkles, limpets and chitons. *Fucus vesiculosis*, one of the most commonly used indicators for radioactivity, is found in locations similar to that of ascophyllym nodosum and also provides food for grazers. *Laminaria* (kelp) is a shallow subtidal flora, whose fast rate of growth and consequent fast biological turnover time makes this an excellent indicator species for radionuclide uptake. The two red algae species, *Chondrus crispus* and *Rhodymenia palmata*, are found in the lower part of the mid-littoral zone and each of these algae has potential for commercial use. Carragenin, a thickener and binder for foods used to prepare puddings and ice cream, is extracted from chondrus (irish moss) and rhodymenia (dulse) is widely collected and consumed by local inhabitants of the Point Lepreau region.

The radionuclide results for marine plants collected at intertidal (denoted by prefix I) and subtidal (s) sites indicated in Figure 38 are given in Table 12. Cesium-137 and Co-60 were measured in several samples while other artificially-produced radionuclides were generally below the detection limit. Although a Cs-137 concentration factor of 123 was measured for Rhodymenia palmata, other concentration factors were lower, in the range of 13 to 29, which are in good agreement with literature values ranging from 17 to 240. Ascophyllum growing tips were not enriched in Cs-137 (compared to other portions of the plant) but did contain measurable quantities of Co-60 and Zr-95. Future measurements will be conducted to determine whether this species is, in fact, an effective concentrator of Co-60.

The relatively short-lived radionuclides, Zr-95 and Mn-54, were measured at levels of 1.2-2.0 Bq/kg and 0.5 Bq/kg, respectively, in several algae samples collected in December 1981. The source of these radionuclides was probably the October 1980 nuclear weapons test carried out by the People's Republic of China. Zr-95 and Ru-103 were also detected in dulse samples collected by the New Brunswick Electric Power Commission (1982) in 1981 near Point Lepreau. Algae were the only samples collected in the marine environment in 1981 which contained any of these short-lived, fallout radionuclides, an observation which emphasizes the value of algae as an indicator species. These results also indicate that care must be taken to distinguish between fallout inputs of radioactivity and those from the nuclear power plant.

5.5 Molluscs

Molluscs are filter feeders consuming plankton, detritus and

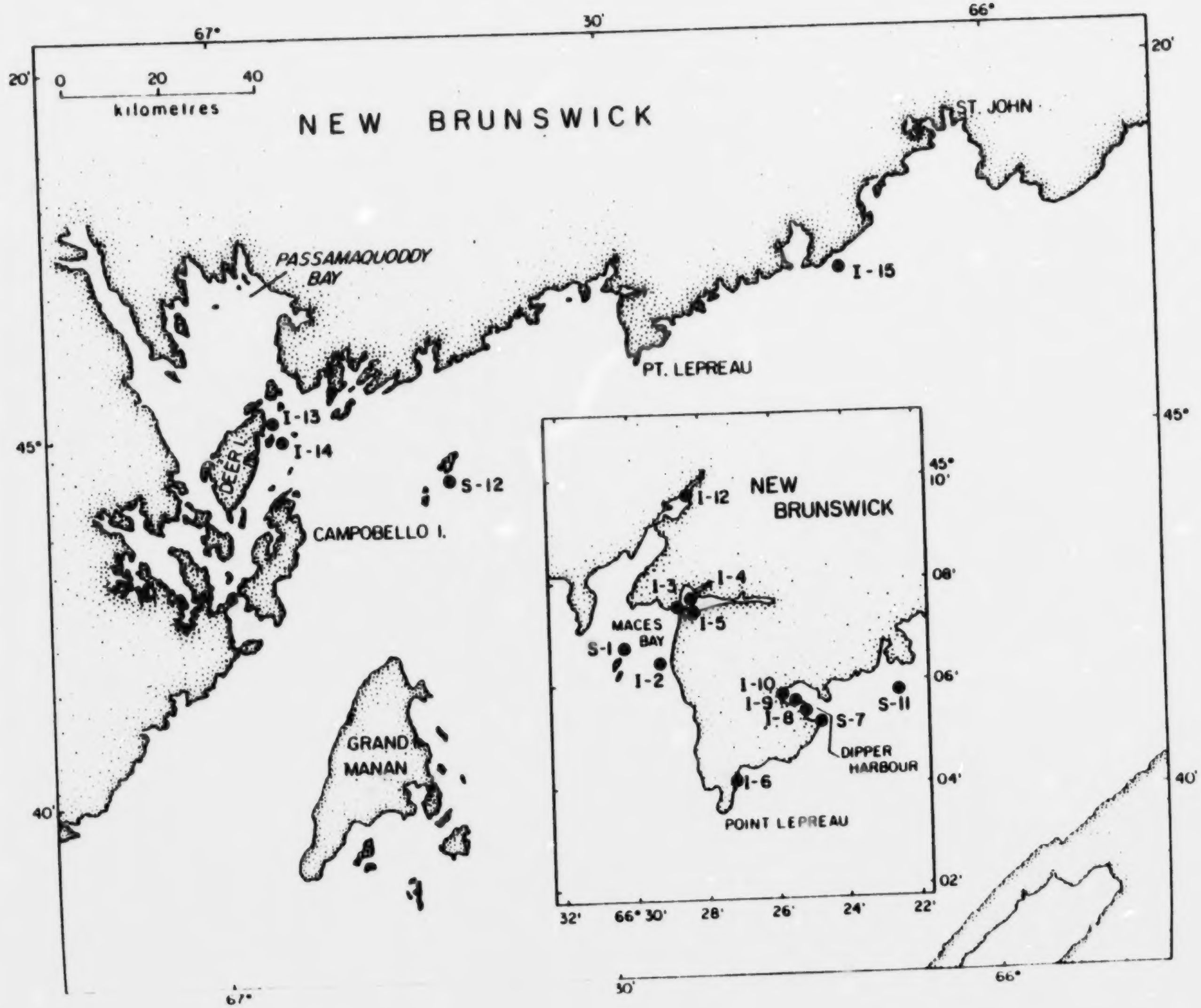


Figure 10. Marine biological sampling stations in Bay of Fundy, including

[illegible] the plants collected near Point Lepreau.

Figure III Marine biological sampling stations in Bay of Fundy include
[illegible]

Table 1. Radionuclide results for marine plants collected near Point Lepreau

Organism	Sample No.	Collection Site	Collection Date	Sample Weight (g)	Geometry	%Water	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)	Other Radionuclides (Bq/kg)
Dulse (*Rhodymenia palmata*)	80-233	I-2	7/2/80	17.3	3	89.3	<2.7	<86	<3.2	-
	80-234	I-8	28/1/80	25.3	3	81.1	2.2 ± 1.3	123	2.2 ± 1.3	-
Irish Moss (*Chondrus crispus*)	80-235	I-8	28/1/80	82.6	2	77.7	<2.0	<134	<1.0	-
	80-236	I-2	7/2/80	47.9	3	87.4	<1.0	<38	<2.0	-
	80-237	I-8	28/1/80	127.9	1	76.2	<0.67	<48	<1.5	-
Kelp (*Laminaria sp.*)	80-238	I-2	7/2/80	64.6	1	86.7	<0.67	<27	<2.0	-
	80-239	I-3	10/1/80	24.7	3	87.8	<3.3	<122	<5.0	-
	80-240	I-9	28/1/80	62.7	1	92.2	1.2 ± 0.7	27	<0.7	-
	81-681	I-6	1/12/81	59.0	3	84.2	0.5 ± 0.3	24	<0.2	-
Rockweed (*Ascophyllum nodosum*)	80-241	I-4	10/1/80	133.2	1	76.9	0.4 ± 0.2	29	<0.7	-
	80-242	I-8	28/1/80	190.7	1	68.8	<0.5	<47	<1.3	-
(Growing Tips)	81-680	I-6	1/12/81	159.3	2	76.4	0.3 ± 0.1	21	3 ± 2	Zr-95 2 ± 1
Rockweed (*Fucus vesiculosus*)	81-682	I-6	1/12/81	157.4	2	76.5	0.18 ± 0.13	13	<0.8	Zr-95 1.2 ± 0.7
	81-683	I-6	1/12/81	43.9	4	74.2	<0.83	<65	<1.2	Mn-54 0.5 ± 0.3
Seaweed (Mixed Species)	81-684	I-6	1/12/81	31.8	4	69.6	<1.5	<137	<1.8	-

suspended particulate organic material and concentrate radionuclides associated with these materials to a marked degree compared to seawater concentrations (Goldberg et al., 1978). Although these radionuclides are generally stored in the hepatopancreas (Schelske, 1973), other organs such as the kidney or excretory organs may also be effective concentrators of radionuclides. Certain types of molluscs have unusually high concentration factors for specific elements. For example, scallops are effective at concentrating Mn in the kidney and oysters have high concentration factors for Zn. Iron is concentrated in the hepatopancreas of oysters from which it is passed to the byssus threads which bind the organism to the substrate (Pentreath, 1980).

Mytilis edulis was used in the Mussel Watch program (Goldberg et al., 1978) as a surveillance organism for marine pollutants and radionuclides. Cesium-137 levels in mussels exposed to nuclear fallout ranged from 0.022 Bq/kg to 0.057 Bq/kg, while those of Pu-239,240 ranged from 2.2 mBq/kg to 5.7 mBq/kg, for samples collected from both the Atlantic and Pacific Ocean coastlines of North America. The bysall threads of mussels were observed to concentrate Pu-239 and Am-241 thirty times more effectively than the soft parts of the mussels. Clams and mussels have also been used extensively for the monitoring of nuclear reactors. Aarkrog (1979) observed elevated levels of Cs-137 (1.0 Bq/kg wet wt.), Ag-110m (4.6 Bq/kg) Zn-65 (2.0-3.7 Bq/kg), Mn-54 (0.5 Bq/kg), Co-60 (1.4-16 Bq/kg) and Co-58 (0.7-6.7 Bq/kg) in *Mytilis edulis* collected in the North Sea near nuclear facilities in Denmark. Blanchard and Kahn (1979) observed similar levels of Cs-137 and Co-60 in clam meat and lower levels of Co-60 (0.6-1.5 Bq/kg wet wt.) and Sr-90 (3.5-15 Bq/kg) in clam shells collected near the Oyster Creek nuclear facility in New Jersey. In laboratory studies of radionuclide uptake by *Mya arenaria*, Harrison (1973) noted that both Cs-137 and Co-60 were concentrated in the "peel" (the non-living material on the exterior of the siphon and the mantle edge), with substantially greater concentration factors observed for the latter radionuclide. Germaine et al. (1979) studied two species of perwinkles, *Littorina sayatilis* and *Littorina obtusata* in the vicinity of the Cap de la Hague nuclear fuel reprocessing plant and found lower levels in the latter species with both species showing activities higher in flesh than in shell. Ru-106 and Ag-110m were detected in all samples while Ce-144, Cs-137 and Sb-125 was detected in most samples.

The following four organisms have been collected at Point Lepreau for radionuclide analyses. (1) *Mya arenaria*. This soft-shelled clam is abundant in most mudflats. It is a filter feeder which consumes plankton. In addition to its commercial value, it is utilized by predatory invertebrates such as the moon snail and green crab as well as flounder, species which in turn lead to higher trophic levels. (2) *Mytilus edulis*. The blue mussel is a common and abundant bivalve found at the low water mark. It is a filter feeder and is consumed by a variety of animals including invertebrate predators, fish and birds. Because of its extensive use in the Mussel Watch program, a considerable quantity of data is available regarding radionuclide levels of this organism. (3) *Modiolus modiolus*. The horse mussel is a subtidal filter feeder whose young are fed upon by invertebrate predators, fish and birds. Adults are probably consumed by starfish. (4) *Littorina littorea*. The common perwinkle is the most common intertidal gastropod. It grazes on seaweed but will also process organic matter deposited on beaches. It is consumed by other invertebrates, fish and birds. Samples of Pecten maximum (scallops) will also be collected in the future as an indicator species for Mn-54.

Mollusc samples collected near Point Lepreau were separated into shell and soft tissues. Individual organs were too small to be easily removed and analysed effectively for their radionuclide content. The results for the molluscs are given in Table 13. Only Cs-137 and Co-60 were detected in these samples, and these radionuclides were present at levels very close to the detection limit. Co-60 levels appear to be slightly lower in the shells compared to the soft tissues. Concentration factors measured for Cs-137 (18-43) are in agreement with literature values for this parameter.

5.6 Marine Crustacea

Crustacea display a remarkable ability to concentrate certain elements and are much more efficient than vertebrates at absorbing radionuclides from food. Some elements such as zinc can be accumulated directly from seawater via the gills. Radionuclides are concentrated in the hepatopancreas, gills, digestive glands and in the shell (Van Weers, 1975; Pentreath, 1980). Manganese-54 activity levels of up to 16 Bq/kg wet wt and Sr-90 levels of up to 21 Bq/kg were measured in crab shell near the Oyster Creek Nuclear Power Station (Blanchard and Kahn, 1975). Crabs collected in the vicinity of the Cap de la Hague reprocessing plant exhibited the highest radionuclide levels in the gills (as great as 6 Bq/kg of Ce-144, 7 Bq/kg of Ru-106 and 0.1 Bq/kg of Cs-137). The two species, Cancer pagurus and Carcinus maenas, contained approximately the same radionuclide levels (Germaine et al., 1979).

Table 13. Radionuclide results for marine crustacea collected near Point Lepreau.

Organism	Sample No.	Organ	Collection		Sample Weight	Geometry	% Water	Cs-137 (Bq/kg)	Concentration Factor	Co-60 (Bq/kg)
			Site	Date						
Sandshrimp (Crangon sp.)	80-312	total	I-3	17/9/80	56.1	2	59.6	0.8 ± 0.5	101	0.8 ± 0.5
Rock Crab (Cancer sp.)	80-314a	flesh	I-6	16/9/80	23.4	3	74.4	<1.2	<90	1.0 ± 0.7
	80-314b	shell	I-6	16/9/80	245.6	1	54.9	<0.3	<45	<0.5
	80-314c	hepatopancreas	I-6	16/9/80	8.6	5	67.4	<2.8	<277	<4.0
Green Crab (Carcinus sp.)	81-685a	flesh	I-6	1/12/81	8.7	5	78.3	<6.8	<326	<8.2
	81-685b	shell	I-6	1/12/81	57.4	3	59.0	<5.2	<451	<8.3
	81-685c	hepatopancreas	I-6	1/12/81	10.5	4	77.9	<9.8	<652	<6.8
	80-313a	flesh	I-8	7/9/80	10.9	4	60.0	<5.0	<600	<8.3
	80-313b	shell	I-8	7/9/80	168.3	1	61.5	<0.8	<96	<0.8
	80-313c	hepatopancreas	I-8	7/9/80	7.2	4	71.3	<10.0	<860	6.7 ± 5.0
Hermit Crab (Pagurus sp.)	80-228a	flesh	S-7	31/1/80	14.7	4	73.5	1.5 ± 1.2	120	<4.0
	80-228b	shell	S-7	31/1/80	34.1	3	61.6	1.7 ± 1.0	192	<2.0
Amphipod (mixed)	81-690	total	I-6	1/12/81	8.7	4	81.0	<8	<456	<4.2

The following crustacea have been collected for analysis in the Point Lepreau monitoring program. (1) Carcinus maenas. The green crab is reported to have arrived in the Bay of Fundy in the 1950s. It is an active predator feeding on intertidal invertebrates including clams and it serves as prey for fish and birds. (2) Cancer sp. The rock crab is a subtidal predator which feeds on various invertebrates and is also prey for fish and birds. (3) Pagurus sp. The hermit crab occupies abandoned shells for protection (often mollusc shells) and also preys on invertebrates. (4) Crangon septemspinosus. The sand shrimp is commonly found in marsh ponds, estuarine embayments and shallow areas with sandy or muddy bottom sediments. It feeds on smaller animals and debris and is consumed by invertebrate predators, fish and birds. (5) Gammarus oceanicus. Amphipods were collected in an intertidal region where they feed on debris and graze on plants.

The radionuclide results measured in crustacea are presented in Table 14. Cesium-137 was detected in crangon and hermit crab muscle and shell. Cobalt-60 was detected in crangon, rock crab flesh and green crab hepatopancreas. Crangon appears to be most suitable for monitoring in a practical sense, since both of the longer lived fallout radionuclides (Cs-137 and Co-60) were easily detectable. The comparatively high measured value of Co-60 (6.7 Bq/kg) in the hepatopancreas is indicative of the high concentration factor of the organ for Co-60.

5.7 Marine Vertebrates

Marine vertebrates occupy an important position in the food web and are of economic importance, but they are not particularly suitable as an indicator species for radionuclide uptake because of their mobility over

Table 14. Radionuclide results for marine molluscs collected near Point Lepreau.

Organism	Sample No.	Organ	Collection		Sample		%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
			Site	Date	Weight (g)	Geometry				
Blue Mussel (*Mytilis edulis*)	80-299a	flesh	I-2	6/2/80	18.7	3	75.9	<3.3	<241	<4.2
	80-229b	shell	I-2	6/2/80	282.8	2	10.5	<0.4	<90	<0.2
	80-230a	flesh	I-5	10/1/80	24.8	3	89.7	<3.2	<98	2.7 ± 1.7
	80-230b	shell	I-5	10/1/80	105.3	3	10.4	<0.6	<179	<0.4
	80-246-53a	flesh	I-15	7-16/2/80	17.6	3	75.4	<5.0	<369	<6.7
	80-254-58a	flesh	I-15	7/2/80	11.3	3	79.6	<3.7	<224	<5.3
	80-259a	flesh	I-15	7/2/80	6.9	4	86.7	<9.8	<392	<12
	80-246-57b	shell	I-15	7-16/2/80	351.6	1	14.5	<0.27	< 68	<0.37
	80-249b	shell	I-15	10/2/80	71.1	3	19.5	<1.0	<242	0.5 ± 0.3
	80-250b	shell	I-15	10/2/80	74.5	3	17.0	<1.0	<249	<1.5
	80-251b	shell	I-15	16/2/80	29.8	3	11.5	<2.7	<708	<3.5
	80-252b	shell	I-15	16/2/80	69.1	3	24.4	<1.0	<227	<0.8
	80-258b	shell	I-15	7/2/80	56.3	3	9.5	<1.2	<317	1.2 ± 0.7
	80-259b	shell	I-15	7/2/80	98.5	3	7.9	<0.8	<230	<1.2
Horse Mussel (*Modiolus modiolus*)	80-231a	flesh	S-1	6/2/80	58.1	1	88.5	<1.0	<35	1.2 ± 0.7
	80-231b	shell	S-1	6/2/80	569.1	1	32.2	<0.07	<14	0.07 ± 0.05
	80-232a	flesh	I-8	10/1/80	42.5	2	83.4	1.7 ± 1.2	83	<1.7
	80-232b	shell	I-8	10/1/80	505.5	1	11.8	0.07 ± 0.05	18	0.07 ± 0.05
	81-688	flesh	I-6	1/12/81	21.0	3	82.7	<3.5	<182	<3.8
	81-689	shell	I-6	1/12/81	259.7	2	14.3	0.17 ± 0.15	43	0.32 ± 0.20
Soft Shelled Clam (*Mya arenaria*)	80-226a	flesh	I-5	10/1/80	82.3	1	87.6	0.5 ± 0.2	25	<0.7
	80-226b	shell	I-5	10/1/80	193.9	1	11.1	<0.3	<133	<0.7
	80-227a	flesh	I-10	6/2/80	24.2	3	83.5	2.5 ± 1.7	124	<2.2
	80-227b	shell	I-10	6/2/80	312.5	1	26.0	0.7 ± 0.2	148	<0.5
Periwinkle (*Littorina littorea*)	81-686	flesh	I-6	1/12/81	9.7	3	73.4	<4.5	<359	2.8 ± 2.5
	81-687	shell	I-6	1/12/81	358.5	2	3.6	<0.3	<96	<0.2

a comparatively wide range. Radionuclide uptake by marine vertebrates occurs by sorption of radionuclides from food or to other material passed through the gut and by direct adsorption of radionuclides from solution. The relative importance of these two pathways varies is a function of feeding habits, radionuclide distributions, etc.

A wide range of radionuclides has been detected in the various tissues of fish exposed both to fallout radioactivity and to radionuclides released from localized sources such as nuclear reactors. Extensive monitoring of fish was conducted at Bikini Atoll, Enitwetok and Christmas Island following the series of nuclear tests carried out at these locations beginning in 1952. Most of the radioactivity resided in the gut contents, indicating that, in general, the accumulation of radioactivity in fish is inefficient compared to other marine organisms.

Zirconium-95 and Ce-141, Ce-144 are among those radionuclides which are poorly assimilated from gut material in fish. Ruthenium-103,106 are more readily accumulated with the highest activities found in the liver, kidney spleen, bone and gills. In contrast to most radionuclides, Cs-137 is concentrated in fish muscle with levels in the range of 1.5-3.0 Bq/kg having been reported for samples exposed only to nuclear fallout (Blanchard and Kohn, 1975; Pentreath, 1977 and Preston, 1970). Strontium-90 tends to follow calcium and is generally concentrated in bone tissues. Iodine-131 was detected in tuna shortly after nuclear tests at Christmas Islands and activities were reported to be 8 to 10 times higher in muscle compared to liver (Palumbo et al., 1966).

Neutron activation products, many of which are isotopes of essential trace elements, are often concentrated to a greater extent than are the fission products. The highest concentration effects occur in organs such as liver, spleen, kidney, gall bladder, and the heart. Zinc-65 is one of the most effectively concentrated isotopes and appears to be one of the few radionuclides which increases in concentration as it passes up the food chain from phytoplankton to zooplankton to fish (Lowman et al., 1966). High Fe-55 levels have also been measured in fish and these are apparently the result of rapid uptake of this radionuclide in iron-poor surface waters of ocean (Palumbo et al., 1966; Preston, 1970). Radionuclide uptake by marine mammals is similar to that of marine fish. The radionuclides Zn-65, Mn-54, Cs-137, Nb/Zr-95, Ce-141 and Ce-144 were detected in fin whales collected off Oregon near the Columbia River in 1963 at similar levels as mesopelagic fish (Osterberg et al., 1964). The Zn-65 distribution revealed highest levels in the liver (0.27 Bq/kg wet weight) and lowest levels in the adipose tissue (0.007 Bq/kg). Zn-65 activities in muscle were 0.10 Bq/kg. Samuels et al. (1970) measured Cs-137 levels in 7 fin whales collected off Nova Scotia and found highest activities in the muscle (0.4-3.4 Bq/kg wet weight), levels in vital organs such as kidney and liver approximately half those of muscle and lowest levels in adipose tissue (average of 0.4 Bq/kg). Levels of 0.4 to 1.9 Bq/kg for Cs-137 measured for 7 harp seals collected in the Gulf of St. Lawrence (Samuels et al., 1970) were slightly lower than that for fin whales. Muscle from young seals was found to contain higher activities of Cs-137 ($\bar{x}$ = 1.7 Bq/kg) than adult seals ($\bar{x}$ = 1.1 Bq/kg), while mammary gland tissue contained slightly less Cs-137 ($\bar{x}$ = 0.94 Bq/kg).

Marine vertebrates selected for monitoring near Point Lepreau include the following. (1) Phoca vitulina. The Harbour seal is a year around resident which feeds on benthic invertebrates and fish. It is at the top of the food chain and it is known to concentrate various types of pollutants.(2) Phocoena phocoena. The Harbour Porpoise is the common resident toothed whale. Major populations enter the Point Lepreau area during the summer months. This species feeds on benthic and pelagic invertebrates and fish and is also known to effectively concentrate certain pollutants. Radioactivity measurements will be made on this species in the future if a sample is obtained on an "opportunity" basis. (3) Pleudopleuronectes americanus. The winter flounder occurs in most all the major marine habitats and feeds on a vast array of plants and animals which makes it an ideal selection as an indicator species for fish. In addition, the flounder is prey for numerous species of fish, birds and mammals. (4) Clupea harengus. The Atlantic Herring feeds largely on plankton including Euphasid shrimp. It is an important commercial species in the Bay of Fundy and is also prey for many other organisms.

Cesium-137 was distributed quite uniformly throughout the muscle, liver and brain tissues of the Harbour Seal, having a range of 1.2 Bq/kg to 2.0 Bq/kg (Table 15). The Cs-137 levels were substantially lower (0.3 Bq/kg) in adipose (fat) tissue. These levels agree well with those reported in harp seal by Samuels et al. (1970). Concentration factors for Cs-137 in muscle, liver and brain tissues range from 133 to 193, while the range for adipose tissue is 46 to 68. No Cs-137 was detected in the throat gland, although the small sample size meant that the detection limit was comparatively high. When it was detectable, Co-60 levels were generally of the same order of magnitude as the Cs-137 levels in the same tissues. An unusually high, and as yet unexplained, level of Co-60 (9 Bq/kg) was measured in the seal throat gland.

Cesium-137 levels in herring muscle were similar to those reported in the literature and measured in seal muscle, while levels in other

Table 15. Radionuclide results for marine vertebrates collected near Point Lepreau.

Species	Sample No.	Organ	Collection		Sample		%H₂O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
			Site	Date	Weight	Geometry				
Harbour Seal (Phoca vitulina)	80-310	muscle	I-13	16/7/80	21.9	3	69.8	1.7 ± 1.5	151	<2.0
	80-327	muscle	I-13	16/7/80	34.1	3	56.1	<1.3	<176	<1.5
	80-243	liver	I-13	16/7/80	155.4	1	67.8	2.0 ± 0.3	193	1.0 ± 0.2
	80-311	liver	I-13	16/7/80	127.8	2	70.5	1.5 ± 0.3	133	<0.5
	80-328	liver	I-13	16/7/80	97.7	2	60.9	1.2 ± 0.5	137	<1.2
	80-245	brain	I-13	16/7/80	41.3	2	78.7	2.3 ± 1.3	149	<3.7
	80-308	brain	I-13	16/7/80	22.0	3	77.0	<2.2	<150	3 ± 1
	80-244	adipose (fat)	I-13	16/7/80	107.2	2	10.9	<1.2	<312	<1.5
	80-309	adipose	I-13	16/7/80	201.5	2	54.3	0.3 ± 0.2	46	<2.0
	80-326	adipose	I-13	16/7/80	236.3	2	32.4	0.3 ± 0.2	68	<0.7
	80-307 80-325	throat gland	I-13	16/7/80	6.2	4	69.3	<4.3	<399	9 ± 5
Herring (Clupea harengus)	80-323a	muscle	S-12	30/10/80	99.6	1	69.7	1.2 ± 0.5	106	<1.8
	80-323a	muscle	S-12	30/10/80	100.3	2	69.3	1.0 ± 0.7	92	<1.8
	80-323c	kidney+ liver	S-12	30/10/80	3.7	5	61.0	<7.5	<697	<11
	80-323d	head	S-12	30/10/80	8.8	4	68.5	<7.0	<662	8 ± 4
	80-323e	gill	S-12	30/10/80	7.0	4	72.1	<9.7	<809	<12
	80-323f	bone	S-12	30/10/80	6.3	4	63.3	<11	<1200	8 ± 5
	80-323g	gut	S-12	30/10/80	24.2	4	54.5	<3.0	<410	<4.0
Winter Flounder (Pseudopleuronectes americanus)	80-315	muscle	S-11	30/10/80	45.2	2	72.9	<1.5	<122	<12

tissues of herring were below the detection limit. Rather high levels of Co-60 were measured in the heads and bones of herrings. Again, it is not clear why levels of this radionuclide are much higher in herring and seal tissues than in plants, animals and physical media in other compartments of the marine environment. No artificial radioactivity was detected in flounder, possibly because of the comparatively small sample size.

The principal conclusions drawn from this first set of pre-operational monitoring results for marine organisms are that, (1) fallout radionuclide levels are low in organisms near Point Lepreau and in most cases are close to the detection limit, and (2) there is no apparent increase in fallout activity levels with increasing trophic level (Figure 39) of the organism. The low radioactivity levels measured in marine organisms are clearly due to the enormous diluting effect of the seawater reservoir.

SAMPLE	Cs-137 (Bq/kg)	Co-60(Bq/kg)	Concentration Factor (Cs-137)
Plankton			
"phyto"	<2.0 - 2.3	<72	<18 - 23
"zoo"	<12	<15	<121
Algae	0.2 - 2.2	<0.2 - 3	13 - 123
Mollusc			
Shell	0.07- 0.7	0.07 - 0.3	18 - 148
Flesh	0.5 - 2.5	<0.7 - 2.8	25 - 124
Crustacea			
Shell	<0.3 - 1.7	<0.5 - 0.8	<45 - 192
Flesh	<1.2 - 1.5	0.8 - 1.0	<90 - 120
Seal			
Muscle	<1.3 - 1.7	<2.0	151 - <176
Liver	1.2 - 2.0	<1.2	133 - 193
Brain	<2.2 - 2.3	<3.7	<150 - 149
Adipose	0.3	<2.0	46 - 68
Throat gland	<4.3	9	<399
Herring			
Muscle	1.0 - 1.2	<1.8	92 - 106
Kidney+ Liver	<7.5	<11	<697
Flounder			
Muscle	<1.5	<3.2	<122

Figure 39. Summary of Cs-137 and Co-60 measurements and detection limits for marine organisms near Point Lepreau.

6.0 TERRESTRIAL AND AQUATIC ENVIRONMENTAL PHASES

When radionuclides are released to the terrestrial environment they become spatially dispersed in abiotic media, principally air and water, and are subsequently channeled into various components of the ecosystem, as illustrated in Figure 40. Upon entering the water column or airstream, radionuclides are dispersed by molecular or turbulent diffusion, processes which generally act to reduce the radionuclide concentration as a function of distance from the source. At the same time the radionuclides are subject to depositional phenonema, which may include gravitational settling, precipitation scavenging, impaction and chemical adsorption and exchange. Adsorption of radionuclides to surfaces is a rapid process for many elements and is generally a function of the surface area and its physico-chemical characteristics. Surface sorption phenonema provide the principal mechanisms which control both the removal of radionuclides from the air-water continuum and their subsequent availability for biological uptake.

Radionuclides in abiotic phases of the environment can enter plants, which are at the base of the grazing food chain, by uptake from soil, deposition from air or sorption from the water. The uptake of radionuclides by herbivores occurs mainly by ingestion of plants and organic detritus (in addition to associated soils and sediments) and by inhalation of aerosol particles, the latter being a particularly important pathway for some insoluable radionuclides. Ingestion of herbivores by carnivores (including, of course, humans) provides one of the final linkages controlling the distribution of radioactivity in the terrestrial environment.

Recycling of radioactivity back into the organic detritus reservoir occurs as the result of excretions, secretions and the death of plants

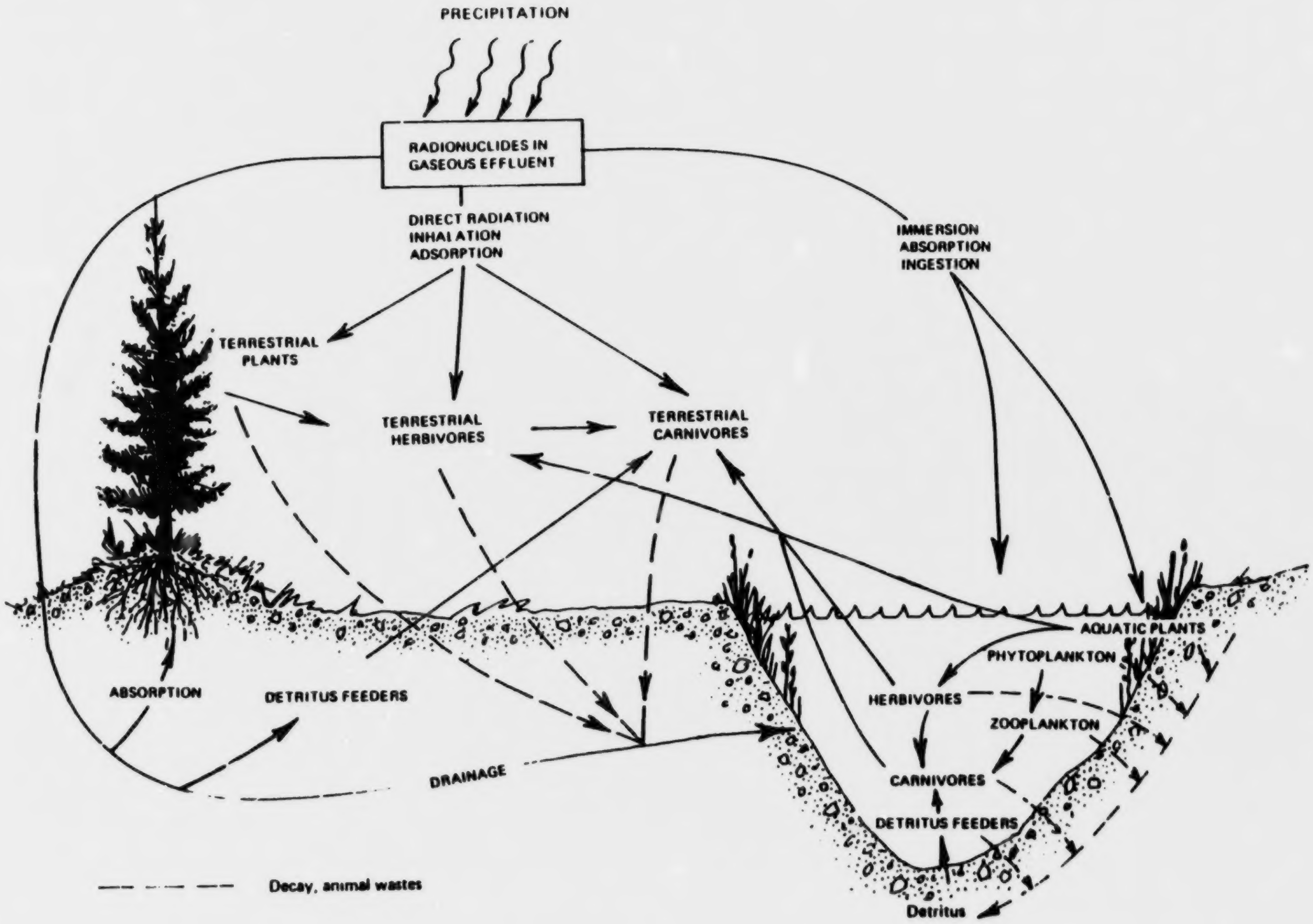


Figure 40. Illustration of transport pathways for radionuclides in the terrestrial and aquatic environments.

and animals. The decay of detritus, promoted by microorganisms, may result in resolubilization of radionuclides and subsequent leaching from soils or dispersion in the water column in particulate form where they can re-enter the food chain. Radionuclides ultimately tend to accumulate in "sinks" composed of the deeper strata of soils, sediments and their associated deposits of organic detritus where radionuclides are tightly bound and have negligible contact with biological compartments. Each step in the food chain provides some form of discrimination, with radionuclides being channeled into specific tissues depending on their physico-chemical properties. The environmental distribution of radioactivity is as much a function of physiological partitioning within the biotic compartments as it is a function of the physical and chemical interactions between the biotic and abiotic compartments. The matrix of rates at which radionuclides move between the various compartments of the ecosystem determine both the final distribution pattern of radioactivity and the speed at which a steady-state distribution is attained. The factors governing these processes will be discussed in the context of the baseline experimental results for Point Lepreau.

6.1 Aquatic Systems

The behavior of radionuclides in aquatic systems is complex because of the large number of competing physical and biological processes which control radionuclide movement. However, since most aquatic systems have well-defined boundaries and a central aqueous compartment with which all other components are in contact, these systems are more suitable for quantitative studies and modeling than most other types of ecosystems. Much has been learned about the cycling of nutrients and metals through aquatic ecosystems through studies of the long-lived, biologically important, fallout radionuclides, Cs-137 and Sr-90. Pendleton and Hanson (1958) conducted an experiment during which an artificial pond was acutely contaminated with Cs-137 and various components of the system were subsequently analyzed for this radionuclide. These authors noted that, although uptake rates for Cs-137 varied considerably throughout the system, all components of the ecosystem eventually attained a steady state concentration, with non-aqueous components having a significantly greater concentration compared to the water. In these types of experiments it has been observed that concentration factors for Cs-137, in addition to most other radionuclides, can be many orders of magnitude for biological materials in freshwater systems, and values of from 100 to 25,000 have been reported for Cs-137 in the literature (Polikarpov, 1966). A notable feature of these concentration factors is the large variability which occurs between different radionuclides, organisms and between separate ecosystems. For example, concentration factors of 0.5 and 154 were measured for tritium and Cs-137, respectively in the same species of alga, Chlorella pyrenoidosa (Weinberger and Porter, 1954; Williams and Swanson, 1958). In fact, concentration factors of up to 300,000 have been measured for P-32 in sessile algae (Kornberg and Davis, 1966). Similar, but much more detailed results were obtained in lake radiotracer experiments carried out in northwestern Ontario in the 1970's (Hesslein et al., 1980; Schindler et al., 1980).

Differences in concentration factors occur between different trophic levels, but the variability is too great to provide a strong basis for generalizations. It has been noted, however, that the ratio, Cs/K tends to increase with increasing trophic level (Pendleton, 1959). It has also been observed that concentration factors tend to be greater in aquatic systems having lower dissolved nutrient concentrations. Templeton and Brown (1963) measured an inverse correlation between Sr-90 concentration factors in brown trout (Salmo trutta) and Ca^{++} in the water at various locations in the U.K., while Kolehmainen et al. (1968) reported decreasing concentrations of Cs-137 in pike (Esox lucius) flesh with increasing K^+ concentrations in finnish lakes. In general, lakes which are oligotrophic (low nutrient levels, comparatively high dissolved oxygen concentrations) in nature are inhabited by organisms having reduced radionuclide concentrations while the opposite is usually the case for eutrophic lakes (Kolehmainen et al., 1968).

Five sampling sites (T-1 - T-5; Figure 41) have been selected in the vicinity of Point Lepreau for the collection of aquatic samples. Two of these sites (T-2, T-4) correspond to small freshwater lakes and the remaining three sites correspond to stream locations. The experimental methodology for the preparation of biological, sediment and water samples and subsequent analysis using nuclear instrumentation are similar to procedures reported earlier in this report. The principal difference is that analytical problems associated with the salt content of the marine samples are not encountered in the present case.

6.1.1 Water Column Results

Cesium-137 is tightly bound to interstitial clay lattice sites in freshwater systems and tends to follow inorganic and organic particle pathways through aquatic systems. The overall partitioning of Cs-137 between water, particles and organic material is a function of both the nature and size of the watershed and the physico-chemical characteristics of the aquatic system (Smith and Ellis, 1982). Cesium-137 levels in the Winnipeg

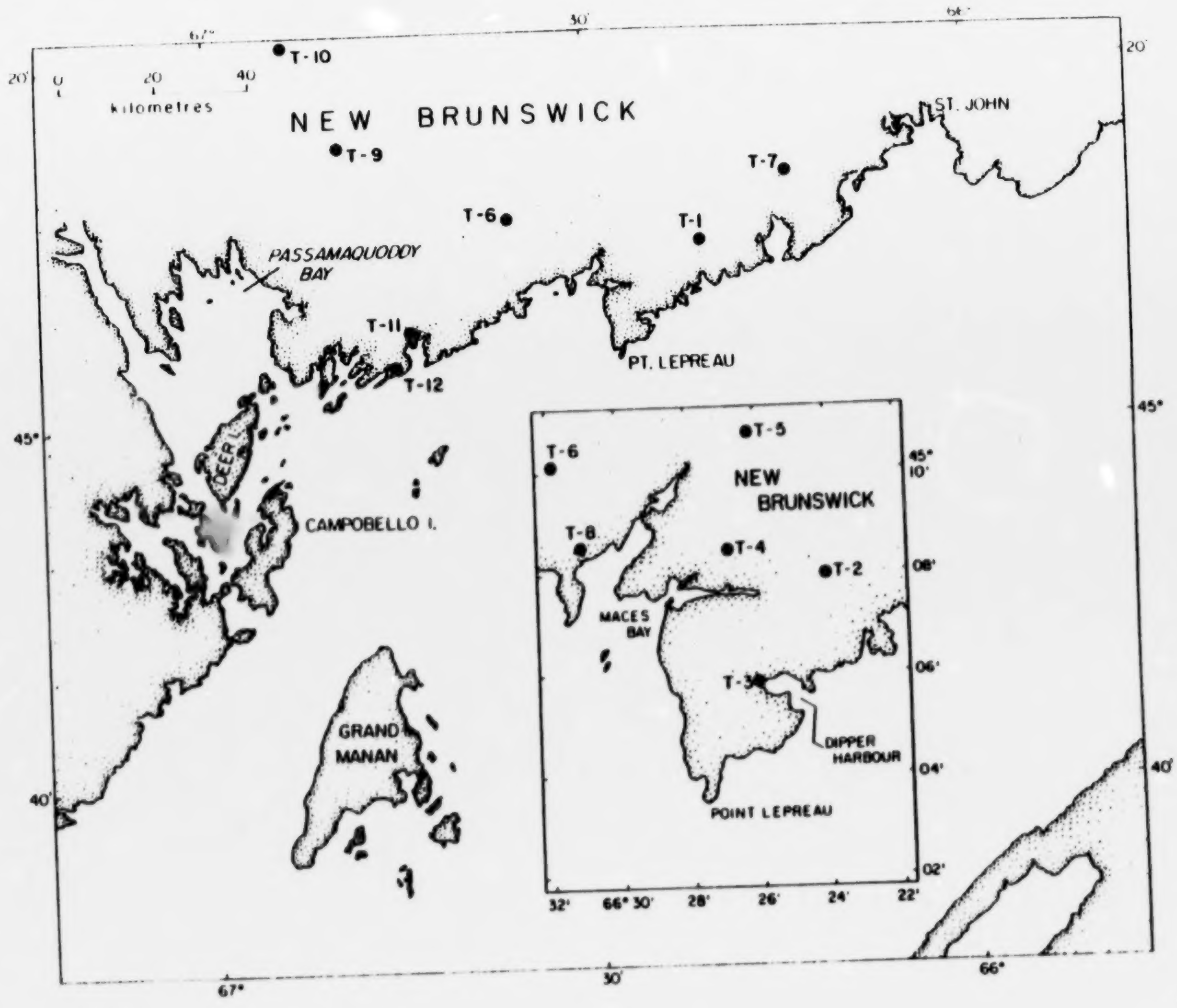


Figure 41. Terrestrial and aquatic sampling sites in vicinity of Point Lepreau.

River in the 1960's and 1970's generally lay in the range of 5 - 10 mBq/l (Brunskill, pers. comm.). Cesium-137 levels in the five freshwater regimes near Point Lepreau lie in the range of less than 1.2 mBq/l to 5.5 mBq/l (Table 16). There is no clear distinction to be drawn between Cs-137 concentrations in the lake (T-2, T-4) and the stream (T-1, T-3, T-5) sampling locations, an observation which is not surprising because the watersheds are similar throughout this region. The precision of the experimental results appears to be good in view of the excellent agreement in the total Cs-137 activity (Table 16) in the three replicate samples collected in Retreat Lake (T-2).

The total Cs-137 activities in these aquatic systems lie in approximately the same range as Cs-137 activities in the Bay of Fundy, although there is much greater variability in the freshwater data. However, in the aquatic systems, almost approximately half of the Cs-137 is associated with particulate material (material retained on a 0.3 μm filter) while in the marine system less than 10% of the Cs-137 is bound to particles. The partioning of a radionuclide between dissolved and particulate forms can be characterized by a distribution coefficient, which is identical to a concentration factor for particulate material. The experimental distribution coefficients for Cs-137 for the Point Lepreau aquatic systems lie in the range of 0.3×10^6 to 1.8×10^6, which is a range typical of aquatic systems (Alberts and Wahlgren, 1981, D. Nelson, pers. comm). The Cs-137 inventory in these systems includes components derived from the erosion of soils from the watershed and resuspended bottom sediments, in addition to Cs-137 derived from directly from atmospheric fallout into the water column (Smith and Ellis, 1982).

Table 16. Cesium-137 and H-3 results for water and suspended particulate material from aquatic sites near Point Lepreau.

Sample #	Sampling Site	Suspended Matter Level (mg/1)	H-3 (Bq/1)	Cs-137 Solution (mBq/1)	Particles (mBq/1)	Particles (Bq/g)	Total Cs-137 (mBq/1)	Distribution Coefficient (K_d) (Bq/g/Bq/ml)
81-04642	T-2	0.97	4.2 ± 1.2	1.7 ± 0.3	2.8 ± 1.3	2.9 ± 1.6	4.5 ± 2.3	1.7×10^6
81-04643	T-2	0.74	-	2.3 ± 0.5	3.2 ± 1.5	4.1 ± 2.1	5.5 ± 2.8	1.8×10^6
81-04644	T-2	1.53	-	2.0 ± 1.0	2.2 ± 1.0	1.6 ± 0.7	4.2 ± 2.4	0.73×10^6
81-04645	T-3	0.61	6.0 ± 1.2	<0.8	0.4 ± 0.6	0.7 ± 1.0	<1.2	$>0.8 \times 10^6$
81-04646	T-4	1.70	5.3 ± 1.2	<0.8	0.8 ± 1.4	0.5 ± 0.8	<1.6	$>0.6 \times 10^6$
81-04647	T-5	1.04	4.5 ± 1.2	3.3 ± 0.5	1.2 ± 1.0	1.3 ± 0.9	4.5 ± 3.8	0.34×10^6
81-04648	T-1	1.00	6.1 ± 1.3	1.0 ± 0.7	0.5 ± 0.7	0.5 ± 0.7	1.5 ± 2.2	0.53×10^6

Tritium measurements in the five aquatic systems (Table 16) are considerably more uniform than the Cs-137 measurements, all values lying in the range of 4.2 - 6.1 Bq/l. The concentration of tritium in continental waters derived from natural (comogenic) processes has been estimated to be in the range of 0.2 - 0.8 Bq/l (Kaufman and Libby, 1954). Present day tritium levels in the terrestrial environment are dominated by bomb produced tritium with values measured in middle latitudes which are typically of the order of 7 Bq/l (Bogen et al., 1979; Gorman and Wong, 1979; Bergman et al., 1979). Hence, tritium levels in the aquatic systems in the vicinity of Point Lepreau are consistent with recent global averages for this radionuclide in freshwater environments. The mean tritium activity in Point Lepreau aquatic systems of 5.2 ± 1.2 Bq/l is in good agreement with the mean activity of 5.9 ± 3.8 Bq/l measured during 1981 in water vapour collected by the air monitoring stations. This agreement reflects the absence of any significant partitioning of this radionuclide subsequent to its deposition and introduction into the continental hydrological cycle. Significantly elevated levels of this radionuclide in aquatic systems, subsequent to the start-up of the Point Lepreau NGS, should by readily discernable.

6.1.2. Sediment Results

Retreat Lake (T-2) is a small (0.1 km^2), shallow (4 m), artifically constructed lake located 9 km northeast of Point Lepreau. It is one of the closest lakes to the site of the Point Lepreau NGS and it would likely intercept any plume of airborne radionuclides transported by the predominantly southwesterly winds (Table 8) which are particularly prominent during the summer months of each year. Furthermore, the convenient access to this privately owned lake makes this an ideal location for monitoring aquatic levels of radionuclides. Eight sediment cores were collected from four locations in this lake during the summer of 1981. These cores were collected by divers using 8 cm diameter, plastic core barrels. The cores were subsequently transported to the AERU laboratory and subsampled following the usual procedures.

Lead-210, Cs-137 and plutonium sediment-depth distributions measured in two of these cores are illustrated in Figures 42 and 43. The Pb-210 activity profile in core 81-4638 decreases exponentially with increasing sediment depth and approaches a Ra-226 supported background level of approximately 1 dpm/g near a depth of 15 cm. The slope of the Pb-210 profile is consistent with a sedimentation rate of 0.098 cm/yr, as illustrated by the solid line in Figure 42. If this core represented an undisturbed sediment regime and the correct sedimentation rate was of the order of 0.1 cm/yr, then the Cs-137 inventory should be restricted to the upper 3 cm of the core. This interval represents material deposited subsequent to 1952, the date for the initial introduction of Cs-137 into the environment. The occurence of significant quantities of Cs-137 below a depth of 3 cm indicates that substantial sediment mixing has occurred, apparently as the result of bioturbation. In fact, Cs-137 is measurable at a depth interval of 14-15 cm which is the depth at which the excess Pb-210 levels approach background levels. This result indicates that both the Cs-137 and Pb-210 distributions have been significantly altered by bioturbation and that the actual sedimentation rate may be significantly less than the apparent sedimentation rate of 0.1 cm/yr. For the case in which sediment mixing occurs by steady-state exchange of adjacent volume elements of sediment as the result of the activities of organisms distributed

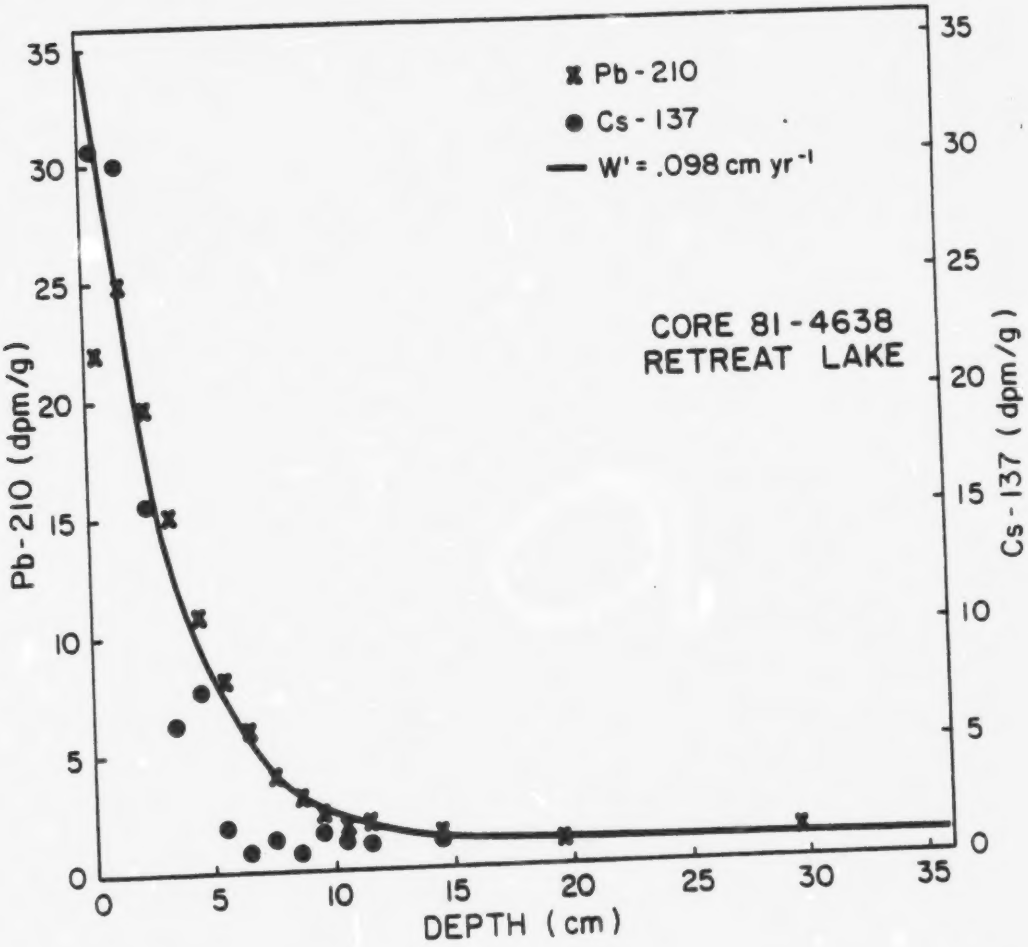


Figure 42. Sediment-depth profiles for Pb-210 and Cs-137 in core 81-4638 from Lake Retreat. Solid curve corresponds to exponential, least squares fit to Pb-210 data for sedimentation rate of 0.098 cm/yr.

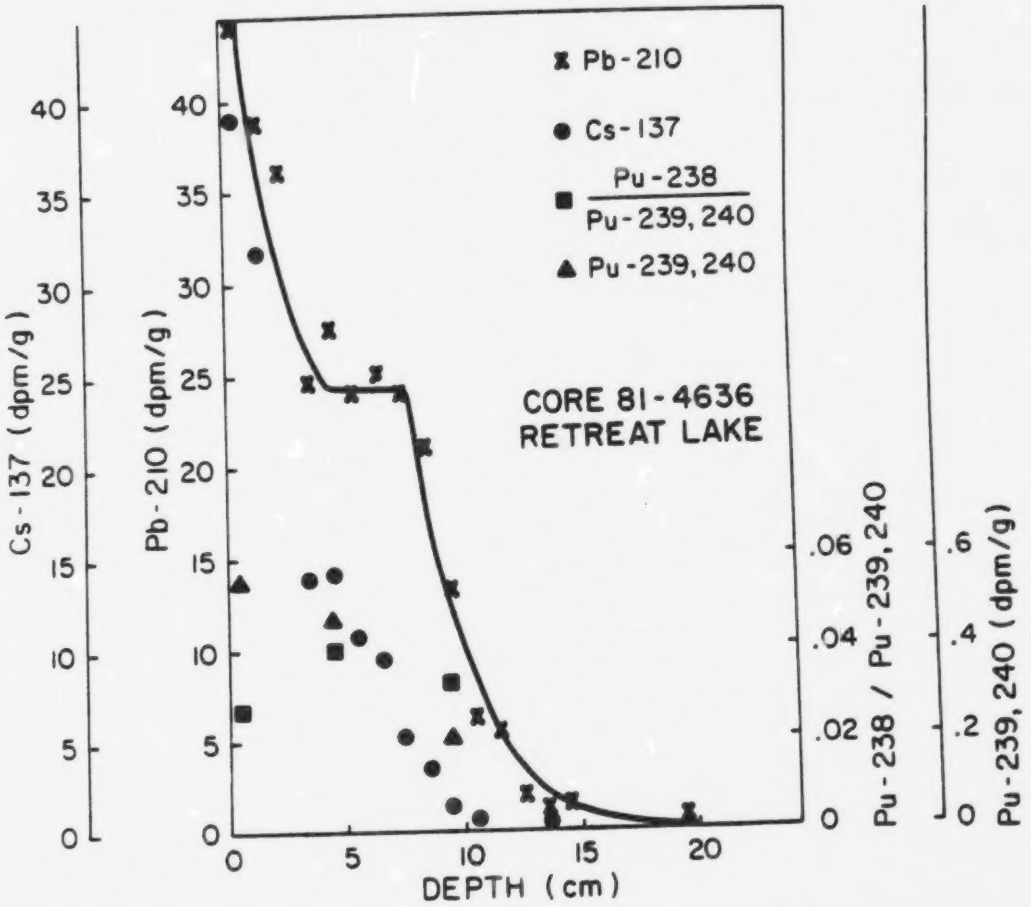


Figure 43. Radionuclide sediment-depth profiles for core 81-4636 from Lake Retreat. Solid curve denotes exponential fit to Pb-210 data including depositional anomaly (slump) between 3 cm and 9 cm.

uniformly throughout the sediment column, the depth distribution of excess Pb-210 ($A_{Pb\text{-}210_{ex}}$), introduced at a constant rate at the sediment surface, may be described by the equation

$$A_{Pb\text{-}210_{ex}} = A^* e^{-(\lambda/K_b)^{1/2} z} \qquad (5)$$

where A^* is the excess Pb-210 activity at the sediment-water interface, λ is the radioactive decay constant (0.031 yr^{-1}) for Pb-210, z is the sediment depth (cm) and K_b (cm^2/yr) is a biological mixing coefficient (Robbins, 1978). This type of mixing model treats the mixing process as analogous to a diffusion process and K_b has the same units as a diffusion coefficient. If the Pb-210 distribution in core 81-4638 was entirely caused by mixing sediment mixing, and the above diffusion analogue model was applicable, then the estimated value of K_b is 0.31 cm^2/yr, a value which is not unusually high for lake sediments. This value is lower by an order of magnitude than particle mixing rates estimated by Krishnaswami et al. (1980) based on Be-7 measurements in Lake Whitney. It may be concluded that the Cs-137 and Pb-210 distributions in core 81-4638 are almost entirely artifacts of sediment mixing phenonema.

The integrated excess Pb-210 inventory in core 81-4638 is 15 dpm/cm^2 and that of Cs-137 is 14 dpm/cm^2 yr, the predicted inventory in a sediment or soil core is 32.3 dpm/cm^2 (Smith and Walton, 1980). Similarly, the predicted inventory of Cs-137 in a soil or sediment core (based on the historical fallout record for New York City) is 26 dpm/cm^2 (Smith and Ellis, 1982). The smaller measured inventories for these two radionuclides in this Retreat Lake core indicate that approximately half of the Pb-210 and Cs-137 deposited in the lake is associated with environmental compartments other than the lake sediments. There is dense mat of vegetation overlying the sediments in Retreat Lake which has a comparatively high Cs-137 activity and much of the Pb-210 and Cs-137 inventory has probably been taken up by this material. Alternatively, particle reactive radionuclides may have been transported from the lake in outflow waters.

Core 81-4636 is characterized by considerably more complex radionuclide distributions compared to the previously-discussed core. The Pb-210 inventory (29 dpm/cm^2) and Cs-137 inventory (16 dpm/cm^2) are both greater than the inventories measured in core 81-4638. Further, the Pb-210 and Cs-137 distributions exhibit a shoulder or plateau between the 3 cm and 8 cm depth interval. This plateau has the appearance of a depositional anomaly which may have been caused by a slumping event (Smith and Walton, 1980). Alternatively, bottom sediment resuspension may have occurred in the past in this portion of the lake as the result of a storm event, followed by the rapid deposition of 4 - 5 cm of material at this location (Robbins and Edgington, 1976; Edgington and Robbins, 1975a,b). If the Pb-210 and Cs-137 inventories between 3 cm and 9 cm are subtracted from the total inventories, the revised radionuclide inventories are in reasonable agreement with those measured in core 81-4638. If this depositional artifact between 3 and 8 cm is discounted, then the resulting Pb-210 sediment-depth profile has an exponentially-decreasing form with a slope similar to that determined for core 81-4638. It may be concluded that the sedimentation and sediment mixing rates at both core locations are probably quite similar, while a localized depositional event has taken place in the vicinity of core 81-4638.

The penetration of both Cs-137 and Pu-239,240 to sediment depths at which the excess Pb-210 activities approach background levels confirms the importance of sediment mixing phenonema in Lake Retreat. The

Pu-239,240/Cs-137 ratios (Table 6) in these freshwater sediments are an order of magnitude smaller than ratios measured in Bay of Fundy sediments, a result which illustrates the much greater efficiency of Cs-137 sorption onto particles and transport to the sediments in aquatic compared to marine systems. The change in the Pu-239,240/Cs-137 ratio with sediment depth is interesting and may reflect some differences in the rates at which the radionuclides are mixed downward into the sediment column as the result of bioturbation. Note that the Pu-238/Pu-239,240 ratios are in good agreement with those measured in the marine sediments. Both sets of data reflect recent (post-1964) inputs of SNAP-9A-derived Pu-238. The uniformity of the Pu-238/Pu-239,249 ratios with sediment depth in core 81-4638 provide additional support for the contention that downward transport of radionuclides is promoted mainly by sediment mixing phenonema.

It may be concluded from the sedimentary radionuclide results that particle-reactive radionuclides released from the Point Lepreau NGS may be efficiently retained in Lake Retreat. The inventories of these radionuclides in Lake Retreat sediments may prove useful as a guide to determining the total fluxes of the radionuclides from the Point Lepreau NGS (using the appropriate scaling factors). However, the geochronology of particle-reactive radionuclide releases will not be preserved in Lake Retreat sediments because of the active sediment mixing which occurs in this system.

6.1.3 Aquatic Plants and Animals

Radionuclide results for aquatic plants and animals collected near Point Lepreau are given in Tables 17 and 18. Again, as in the case of the marine samples, the detection limits for a radionuclide of interest can be

Table 17. Radionuclide results for aquatic plants collected near Point Lepreau

Species	Sample No.	Component	Collection Site	Collection Date	Sample Weight	Sample Geometry	%H_2O	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Zr-95 (Bq/kg)	Ce-144 (Bq/kg)	Be-7 (Bq/kg)	Mn-54 (Bq/kg)
Horse Tail (*Equisetum sp.*)	81-4673	leaves	T-1	14/7/81	39.8	1	76.9	113 ± 7	1.3×10^4	16 ± 8	25 ± 15	113 ± 53	4.7 ± 2.2
	81-4674	leaves	T-2	13/7/81	42.9	1	86.5	230 ± 8	1.6×10^4	10 ± 5	43 ± 17	93 ± 50	<4.2
	81-4675	leaves	T-3	13/7/81	29.1	2	77.6	5 ± 2	560	15 ± 10	48 ± 18	77 ± 43	<5.5
	81-4676	leaves	T-4	13/7/81	28.7	2	74.8	16 ± 5	2.0×10^3	15 ± 10	58 ± 17	93 ± 50	<5.5
	81-4677	leaves	T-5	14/7/81	44.7	2	79.6	8 ± 3	820	<20	48 ± 15	165 ± 47	<5.2
Wild Iris (*Iris versicolor*)	81-4669	total	T-2	14/7/81	31.9	1	82.2	55 ± 7	4.9×10^3	25 ± 25	36 ± 25		
	81-4671a	leaves	T-4	13/7/81	47.4	1	70.9	16 ± 4	2.3×10^3	<22	25 ± 12	11 ± 7	
	81-4671b	roots	T-4	13/7/81	16.1	4		28 ± 8	4.1×10^3	<43	<43	<96	
Water Lily (*Nuphar sp.*	81-4668	total	T-1	14/7/81	61.1	1	82.0	39 ± 3	3.5×10^3	20 ± 10	<37		1.8 ± 1.5
	81-4672a	leaves	T-5	14/7/81	38.2	1	93.3	79 ± 4	2.7×10^3	<26	<37		2.3 ± 1.7
	81-4672b	roots	T-5	14/7/81	9.7	4		30 ± 5	1.0×10^3	<74	<75		<12
	81-4691a	leaves	T-5	14/7/81	44.5	1		85 ± 6	1.7×10^3	<33	<130		<5.5
	81-4691b	roots	T-5	14/7/81	25.8	1	96.1	28 ± 6	550	<45	<5.3		<7.1
Cattail (*Typha sp.*)	81-4670	total	T-3	13/7/81	23.6	1	88.0	<1.3	<80	<41	<57		<6.8
Pond weed (*Pontamogeton*)	81-4678a	leaves	T-1	14/7/81	5.3	4	43.1	<13	$<4 \times 10^3$	<200	<120		<20
	81-46788b	roots	T-1	14/7/81	42.9	2	55.1	32 ± 3	7.2×10^3	<53	<49		<5.5
	81-4688	total	T-4	13/7/81	10.4	4	62.6	23 ± 3	4.3×10^3	66 ± 25	23 ± 11		35 ± 22
Lake vegetation (sediment surface)	81-4649	total	T-2	22/7/81	42.1	1	70.2	28 ± 5	4.2×10^3	<13	<37		<4.6

Table 18. Radionuclide results for terrestrial and aquatic organisms collected near Point Lepreau.

Organism	Sample No.	Organ	Collection Site	Collection Date	Sample Weight (g)	Sample Geometry	$\%H_2O$	Cs-137 (Bq/kg)	Concentration Factor (Cs-137)	Co-60 (Bq/kg)
Muskrat (*Ondatra zibethicus*)	80-297a	liver	T-4	3/3/80	26.53	3	72.4	108 ± 5	1.5×10^4	<4.3
	80-304a	liver	T-5	17/11/80	8.64	5	72.7	53 ± 8	7.2×10^3	<3.3
	80-297b	muscle	T-4	3/3/81	6.84	4	74.0	173 ± 15	2.3×10^4	<6.7
	80-340b	muscle	T-5	17/11/80						
	80-297c	bone	T-4	3/3/81	10.04	5	37.0	5.3 ± 1.5	1.7×10^3	2.2 ± 1.5
	80-304c	bone	T-5	17/11/80						
	80-297d	brain	T-4	3/3/81	1.33	5	72.0	77 ± 25	1.1×10^4	23 ± 12
	80-304d	brain	T-5	17/11/80						
Varying Hare (*Lepus americanus*)	80-305a	liver	T-5	3/3/81	20.42	3	76.3	283 ± 7	3.4×10^4	2.5 ± 2.0
	80-305b	muscle	T-5	3/3/81	5.45	4	79.4	410 ± 17	4.2×10^4	6.7 ± 5.2
	80-305c	brain, lung, kidney	T-5	3/3/81	5.89	4	80.5	80 ± 3	7.8×10^3	<3.3
Meadow Vole (*Microtus pennsylvanicus*)	80-306	muscle, liver intestine	T-5	3/3/81	2.66	5	84.2	37 ± 10	2.9×10^3	7.7 ± 4.3
Frog (*Rana clamitans*)	81-4679	total	T-1	14/7/81	6.53	4	80.0	32 ± 6	3.2×10^3	<13
(*Rana sylvatica*)	81-4685	total	T-3	13/7/81						
(*Rana clamitans*)	81-4687	total	T-4	13/7/81	7.48	4	85.0	49 ± 9	3.7×10^3	<12
(*Rana catesbyiana*)	81-4692	total	T-5	14/7/81						
(*Rana catesbyiana*)	81-4684	total	T-2	13/7/81	13.84	4	78.9	24 ± 3	2.5×10^3	<4.0
Fish (*Cyprinidae* +	81-4682	total	T-2	13/7/81	4.3	5	88.0	12 ± 5	720	<7.0
Pungitius pungitius)	81-4690	total	T-5	14/7/81						
Cyprinidae	81-4689	total	T-4	13/7/81	5.3	4	86.8	5.5 ± 4.0	360	<17
Cyprinidae	81-4680	total	T-1	14/7/81	17.6	3	87.5	17 ± 4	1.1×10^3	<4.7
Insects (peepers)	81-4686	total	T-4	13/7/81	0.20	5	5.5	<143	1.4×10^3	<100
Fresh water invertebrates	81-4681	total	T-2	13/7/81	4.1	5	85.3	19 ± 6	1.4×10^3	5.0 ± 2.7

estimated using Appendices 3 and 4. Concentration factors (C.F.) have been estimated for these samples for Cs-137 using equation 4 (Section 5.1) where the $A_{Cs\text{-}137_{water}}$ is the mean Cs-137 activity measured (in solution) in the freshwater systems near Point Lepreau (Table 16). The interpretative difficulties in drawing quantitative comparisons between concentration factors measured in completely different sampling regimes and the questionable validity of the concentration factor, itself, as a quantitative tool have been outlined in a previous section. However, the concentration factor provides a useful means for making qualitative comparisons of radionuclide uptake by different organisms in the same aquatic systems.

The predominant mechanisms for the accumulation of radionuclides in aquatic plants and invertebrate animals are adsorption and absorption, generally leading to direct equilibration with air and water, while ingestion is the principal mechanism for radionuclide accumulation by vertebrates, particularly for the larger carnivores. This generally results in a faster response time to changes in environmental levels of radioactivity by aquatic plants compared to vertebrates. This prediction is confirmed to some extent by the Point Lepreau results where short-lived fission products (Zr-95, Ce-144) from the 1980 Chinese nuclear weapons test were observed in some aquatic plants, but were all below the detection limit in vertebrates.

One of the most effective indicator species in the aquatic environment appears to be water horsetail (Equisetum sp.). This is an important primary producer utilized by various aquatic insects, invertebrate herbivores and terrestrial herbivores. It grows abundantly along streams and in ponds and lakes in the Point Lepreau region and can be easily collected from mid-spring until late autumn. The Cs-137 activity in horsetail varied considerably between the two samples collected at sites T-1 and T-2 and those collected at the other three sites, with both the Cs-137 activities and concentration factors being an order of magnitude greater in the former samples (Table 17). Cesium-137 activities in the water columns at the different sites do not exhibit this degree of variability. Further, the comparative uniformity in the activities of other fallout radionuclides (Zr-95 and Ce-144) measured in these samples imply that there have been no significant variations in their degree of exposure to recent, airborne inputs of radioactivity. There is some evidence to indicate that the older, more mature water horsetail plants have higher Cs-137 activities and the possibility of long-term of accumulation of this radionuclide in the root systems is a subject of further investigation.

The occurrence of Zr-95, Ce-144 and Mn-54, in addition to the cosmogenic radionuclide, Be-7, in water horsetail is evidence for its efficiency as an aquatic indicator species for airborne inputs of radionuclides. The mean ratio of Zr-95/Ce-144 of 0.32 measured in this plant is in good agreement with the mean value of measured in lichen samples and only slightly less than the value of measured on air particulate samples (Figure 34). In contrast, the ratios of Zr-95/Cs-137 and Ce-144/Cs-137 measured for these plants fall into two categories; values (4675-7) which are substantially greater than those measured on air particulate material and values (#4673-4) which are considerably less. Again, these observations reflect the unusual differences in Cs-137 uptake in different water horsetail plants.

Cattail (Typha sp.), like horsetail, is also an emergent plant. Its very low concentration factor for Cs-137 is consistent with previous measurements summarized by Polikarpov (1966) in which cattail had one of the lowest concentration factors for a range of 20 radionuclides. In contrast, wild iris (Iris vesicolor) has comparatively high and uniform concentration factors for Cs-137, with slightly greater activities measured in the roots compared to the leaves. The detection of Zr-95 and Ce-144 in wild iris indicates that airborne pathways play an important role in radionuclide uptake, a result that would be expected for most emergent plants.

Water lily (Nuphar sp.) is a floating plant and has Cs-137 concentration factors slightly less than those measured for wild iris. In contrast to wild iris, Cs-137 activities are greater in the leaves of water lilies compared to the roots. The Cs-137 activities measured in the present study are similar to values determined by Wahlgren and Marshall (1975) for water lilies collected in Lake Michigan.

Pond weed (Potamogeton) is a submerged vascular plant and would be expected to achieve equilibration with radionuclide activities in the water column. However, the comparatively high levels of Zr-95 and Ce-144 in a sample from the T-4 lake site indicate that this plant either received direct inputs of fission products from the atmosphere, possibly due to its elevation above the water surface, or there are very rapid particulate transport processes for its radionuclide uptake in the water column. The Mn-54 activity in sample #81-4688 is unusually high and further samples of pond weed will be examined from site T-4 to confirm this result. Cesium-137 activities of the order of 20 Bq/kg (dry wt.) have been measured by Wahlgren and Marshall (1975) in samples collected in Lake Michigan, in good agreement with the present results. Surface detritus and vegetation from Lake Retreat (#81-4649) is also high in Cs-137, having a concentration factor close to the value estimated for pond weed. This would be expected in view of the similarities in the habitat and vascular structure of these different species of vegetation.

In general, the Cs-137 activities in aquatic plants lie in the range of 10 - 100 Bq/kg (2 - 20 Bq/kg wet wt.) and the Cs-137 concentration factors tend to lie in the range of $10^3 - 10^4$. Both the Cs-137 activities and concentration factors are lower by several orders of magnitude in marine plants. The more efficient uptake of Cs-137 in freshwater plants reflects the lower ionic strength of aquatic media and the reduced inventory of Group I cations competing with Cs-137 for lattice sites both on particles and in biological systems.

Aquatic animals sampled for radionuclide analyses (Table 18) include several species of frogs, minnows (Cyprinidae), ninespine sticklebacks (Pungitius pungitius) and freshwater invertebrates (several species of worms). Cesium-137 activities in frog samples lie in the range of 24 to 49 Bq/kg, while the Cs-137 concentration factors are more uniform, having values of $2.5 - 3.7 \times 10^3$ Bq/kg. Cesium-137 activities in fish and invertebrates are lower, with the latter organisms exhibiting measurable levels of Co-60. Radionuclide uptake in amphibians and fish tends to proceed by ingestion pathways rather than direct uptake from the water. This has been demonstrated by numerous studies such as those of Davis and Foster (1958) in which Columbia River fish were reported to be 100 times as radioactive as fish maintained in water having the same radionuclide levels but fed uncontaminated food. Ingestion pathways may include the uptake of suspended particulate material or bottom sediment enriched in particle-reactive radionuclides. For example, the cause of high Cs-137 burdens in rainbow trout sampled in a lake in the Colorado mountains was attributed to the ingestion of bottom detritus by the fish as they pursued benthic organisms within the bottom sediments (Gallegos et al., 1971). The size of a fish is also a factor determining its radionuclide content. The actual quantity of radioactivity (per unit weight of fish) tends to decrease with increasing size of the fish, a relationship which is balanced to some extent by the fact that the biological half life tends to increase with increasing size of the fish. These two relationships may offset one another, resulting, in some cases, in no predictable pattern in the variation of radioactivity with size. In the present case, the samples of fish, collected from several different aquatic sampling sites, are sufficiently uniform in Cs-137 activities that the data provides a reasonable baseline for future post-operational aquatic studies at Point Lepreau.

One of the few previous studies of Cs-137 in uptake in a wide range of aquatic organisms was carried out by Pendleton and Hanson (1958) on samples collected from a low-level radioactive waste swamp on the Hanford Reservation containing Cs-137 levels of 4.0 - 200 Bq/l in the water. Cesium-137 concentration factors for fish (carp and sunfish) and frogs measured by Pendleton and Hanson (1958) were similar to those determined in the present study. However, the Cs-137 concentration factors in aquatic plants (including floating, submerged and emergent species) were generally an order of magnitude smaller than those measured at Point Lepreau. Pendleton (1959) concluded from these (and similar measurements) that Cs-137 becomes concentrated to progressively higher levels as it is transferred up the food web. However, the present study indicates that the Cs-137 concentration factors are approximately the same or even greater in aquatic plants compared to aquatic animals and that there is no evidence for concentrating effect for this radionuclide with transfer up the food web. It is difficult to make a direct comparison between the two experiments because of differences in experimental conditions, but it is possible that a true steady-state in the Cs-137 distribution had not been attained in the Hanford waste swamp environment. Williams and Pickering (1961) subsequently reported results indicating that Cs-137 concentration factors decrease as the trophic level increases from water to carnivorous fish, a view that now appears to be generally accepted (Davis, 1963).

6.2 Terrestrial Systems

Greater variability in radionuclide levels is generally observed in terrestrial compared to aquatic systems. The residence times of most radionuclides in air are much shorter than their residence times in water and, as a result, a steady-state in radionuclide levels can be established throughout the different components of an aquatic system more rapidly and efficiently than throughout a terrestrial ecosystem. Physiography, microterrain and microclimate are important factors governing the nature of terrestrial ecosystems, their driving variables and the properties of radionuclide transport within these systems. Biological uptake of fallout radionuclides is generally more efficient in the biotic components of terrestrial environment, compared to aquatic and marine ecosystems, because of the absense of a diluting medium such as water. This results in comparatively high levels of fallout radionuclides in land-based plants and animals as is shown below by baseline measurements conducted on samples collected at Point Lepreau.

6.2.1 Soil Phases

Soils are composed of a parent material, the underlying geologic or mineral substrate, and an organic increment in which organisms and their products are intermingled with the finely divided particles of the modified or weathered parent material. The texture and porosity of the soil, in addition to its chemical composition, largely determine the availability of nutrients to plants and soil animals. Soil profiles are composed of a sequence of three principal horizons from the surface downward. The upper "A" horizon (top soil) is an organic rich zone composed of the bodies of plants and animals in various stages of decomposition. The underlying "B" horizon consists of mineral soil in which organic material has been converted to inorganic material through the process of mineralization. The third "C" horizon represents the comparatively unmodified parent material which may correspond to the original mineral formation or to material transported to the site by glaciers, wind, etc. The soil profiles and relative thickness of each horizon are generally a function of climate and topography.

The upper layers of soils are fairly efficient long-term collectors of many radionuclides supplied by atmospheric precipitation (Hardy et al., 1973; Nozaki et al., 1978; Dodds and Matthews, 1976). However, comparison of soil radionuclide contents is complicated by the diversity of soil composition, mainly because of the many parent materials available and the varied conditions for their degradation to soil. The uppermost soil layer, in particular, is continually modified by the effects of cultivation, plant growth, animal habits, erosion, dust deposition, etc. Even in the absense of significant soil disturbance, ground water percolation provides a mechanism for leaching radionuclides from soil particles, resulting in significant differences in radionuclide concentrations even within soils of the same type. These factors combine to make soils a less effective compartment, compared to sediments, for resolving the temporal record of radionuclides inputs to aquatic and marine systems.

Soil profiles were collected at two sites near Point Lepreau by excavating a shallow trench, 25 cm deep, and subsampling down the exposed soil face. Layers of soil from each 2 cm depth interval, approximately 20 cm x 20 cm in surface area, were collected from the face of the trench. These soil samples were returned to the laboratory, freeze-dried and analysed for Cs-137 using a GeLi detector. From the % water loss and a knowledge of the volume of soil in each sample, the solids density was determined as a function of soil depth. These data can be used to estimate the total inventory of Cs-137 in each soil column.

Most radionuclides, and Cs-137 in particular, bind rather tenaciously to litter, humus and small inorganic particles such as clay. They do not tend to move into the deeper soil layers and are generally retained in the upper litter-humus or "A" horizon. The Cs-137 soil-depth profiles (Figure 44) for the two cores collected in this study are consistent with these general observations.

One core was collected in an inclined grassy field which sloped downward to Retreat Lake (terrestrial site, T-2). The second core was collected at the base of grassy, shallow depression at the very tip of Point Lepreau, 1 km south of the reactor site. In both cores, most (92% - 94%) of the Cs-137 resides in within the organic-rich "A" horizon of the soil, with a small component having penetrated into the mineralized "B" zone. The comparatively high and uniform distributions of Cs-137 within the upper 8 cm of the soil profiles indicates that significant reworking of these soils (activities of organisms, plant root penetration, etc.) has occurred during the recent past. However, the two cores differ in terms of their total inventory of Cs-137, with the inventory at the Pt. Lepreau site (19 dpm cm^{-2}) being somewhat higher than that of the inventory (14 dpm cm^{-2}) at

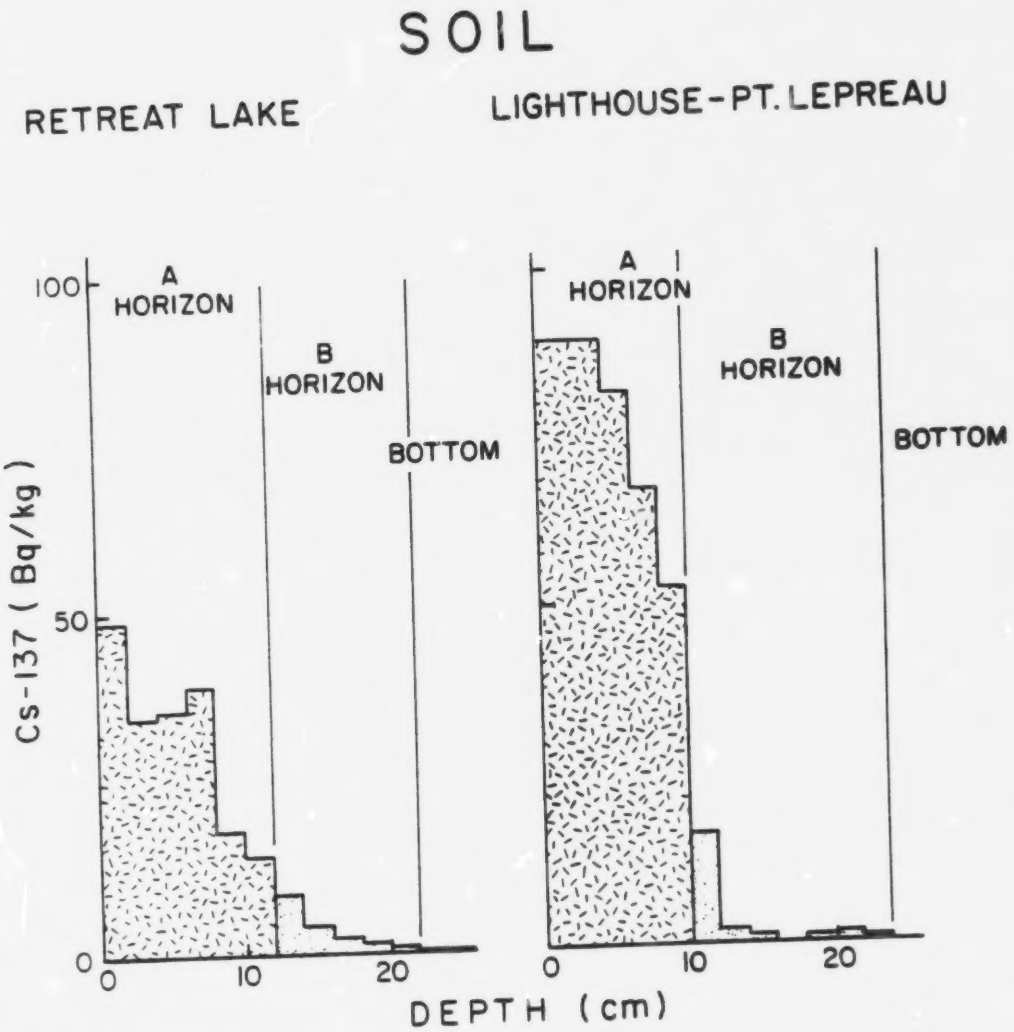


Figure 44. Cs-137 soil-depth profiles for soil cores collected near Retreat Lake and the tip of Point Lepreau. Most of the fallout Cs-137 is confined to the organic-rich, "A" horizon of the soil.

the Retreat Lake site. The inclined topography at the Lake Retreat site and the proximity of a sizable water shed suggests that some of Cs-137 may have been transported from the soil site both by soil erosion processes and by leaching of Cs-137 from the soil itself. In contrast, the site of the Point Lepreau soil core favours the accumulation of Cs-137, acting as a small water shed for the surrounding terrain. Hence, the different Cs-137 inventories of the two soil cores are consistent with the topography of their sites.

The Cs-137 inventories at both locations are less than the inventory (26 dpm cm^{-2}) which would be expected from weapons fallout alone. Some of the Cs-137 is bound up in the overlying grasses and other vegetation, but this quantity should be largely restored to the soil-litter carpet during the winter months. The reduced Cs-137 soil inventories are probably due to net erosional removal of particle-associated Cs-137 or to the release of Cs-137 from the soils by leaching processes. These results indicate that soils in the vicinity of Point Lepreau will retain a significant fraction of particle-reactive radionuclides and soil measurements of these radionuclides will be useful as indicators of reactor-released radionuclides. However, in view of the uncertainties associated with the interpretation of soil profiles, these measurements cannot by themselves be used as a quantitative index of the total flux of radionuclides from the reactor.

6.2.2. Terrestrial Plants

Radionuclides introduced to terrestrial systems generally become associated with plants by uptake from soil, by direct deposition upon the plant surfaces or by sorption from water. Radionuclides usually enter plants from soil solution in ionic form, are drawn through the root membrane and into plant tissues where translocation or chemical incorporation into the tissues can occur. Radionuclides attached to plant surfaces may be assimilated into plant tissues through stomates or epidermal tissues. Much of the radioactivity associated with plants may be simply sorbed onto the plant surfaces and is subsequently removed by leaching or washing (Takashima et al., 1978). Plants which have been selected for radioactivity monitoring at Point Lepreau include lichen, moss, blueberry plants and alders. Radionuclide results for samples collected at the sites indicated in Figure 41 are given in Tables 19-21.

Lichen

Lichens have displayed a remarkable ability to absorb and retain radionuclides. This ability is related to the morphological and physiological characteristics of lichens which have evolved to enable these organisms to live in extreme environments and to obtain a large proportion of their nutrients directly from atmospheric precipitation. Lichens are generally mat-like and cover a large amount of ground area relative to their biomass. Branching and the scabrous, flattened thalli of lichen produce a high surface area/weight ratio. The large surface area, efficient ground coverage and low profile, all combine to allow lichens to intercept a large proportion of the fallout debris deposited as dust or precipitation. Once radionuclides are taken up in the lichen thallus, they are lost at a slow rate compared to their loss rates from other forms of vegetation, apparently because the lichen has evolved efficient mechanisms to conserve and retain nutrients that may be in scarce supply. The effective retention time for Cs-137 in lichens as been estimated to be greater than 10 years

Table 19. Radionuclide results for lichen samples collected in southern New Brunswick.

Species	Sample No.	Collection Site	Collection Date	Sample Weight	Geometry	%H_2O	Cs-137 (Bq/kg)	Be-7 (Bq/kg)	Ce-144 (Bq/kg)	Ce-141 (Bq/kg)	Ru-103 (Bq/kg)	Ru-106 (Bq/kg)	Zr-95 (Bq/kg)	Nb-95 (Bq/kg)	Mn-54 (Bq/kg)
Lichen	80-280	T-1	15/7/80	113.2	1	6.2	297 ± 7	-	43 ± 13	-	-	<43	<60	-	<3.5
(*C. rangiferina*)	80-284	T-2	15/7/80	110.5	1	6.0	222 ± 5	-	45 ± 13	-	-	<48	<73	-	<4.0
	80-290	T-3	15/7/80	119.9	1	4.6	95 ± 3	-	<33	-	-	<40	<50	-	<4.3
	80-296	T-4	7/7/80	100.3	1	4.8	185 ± 5	-	28 ± 12	-	-	<38	<80	-	<4.8
	80-303	T-6	7/7/80	103.4	1	4.6	363 ± 7	-	60 ± 20	-	-	<50	<88	-	<3.5
	81-4650	T-4	14/7/81	81.6	1	9.3	335 ± 8	583 ± 67	410 ± 33	32 ± 10	13 ± 8	<53	117 ± 8	225 ± 15	8 ± 2
	81-4651	T-3	14/7/81	95.2	1	8.8	328 ± 8	500 ± 80	322 ± 35	23 ± 12	20 ± 8	<40	93 ± 8	178 ± 19	6 ± 2
	81-4652	T-2	14/7/81	90.2	1	8.2	112 ± 5	167 ± 33	142 ± 20	9 ± 4	11 ± 5	32 ± 10	52 ± 7	88 ± 14	5 ± 2
	81-4653	T-1	14/7/81	90.4	1	8.5	156 ± 5	333 + 50	208 ± 22	<20	15 ± 6	58 ± 20	62 ± 7	133 ± 13	4 ± 2
	81-4654	T-6	14/7/81	91.2	1	8.5	235 ± 7	550 ± 50	485 ± 32	28 ± 8	20 ± 7	117 ± 28	113 ± 8	256 ± 14	8 ± 2
	81-4655	T-7	14/7/81	77.8	1	9.0	285 ± 7	500 ± 67	333 ± 33	32 ± 10	17 ± 7	78 ± 28	97 ± 8	195 ± 18	8 ± 2
	81-4656	T-8	14/7/81	81.5	1	7.9	228 ± 7	417 ± 50	257 ± 27	14 ± 4	13 ± 6	53 ± 30	72 ± 8	157 ± 14	6 ± 2
	81-4657	T-9	14/7/81	100.1	1	9.5	167 ± 5	383 ± 33	252 ± 22	18 ± 7	13 ± 5	70 ± 22	73 ± 5	141 ± 9	6 ± 1
	81-4658	T-10	14/7/81	100.5	1	8.3	145 ± 5	283 ± 50	218 ± 20	13 ± 4	11 ± 6	<25	70 ± 7	131 ± 14	4 ± 2
	81-4659	T-11	14/7/81	73.4	1	8.5	120 ± 5	300 ± 33	200 ± 23	13 ± 7	<20	<30	50 ± 7	98 ± 10	<7
(*Cladina arbuscula*)	81-4665	T-7	14/7/81	45.4	1	11.7	265 ± 10	417 ± 50	393 ± 55	42 ± 23	<20	98 ± 48	105 ± 17	212 ± 42	2 ± 1
Bog Moss	80-275	T-1	1/12/80	38.7	1	75.0	30 ± 7	-	<73	-	-	<70	<55	-	<7
(*Sphagnum sp.*)	80-282	T-2	3/12/80	39.7	1	74.5	93 ± 7	-	<70	-	-	<62	<50	-	<7
	80-285	T-3	1/12/80	32.0	1	79.7	58 ± 7	210 ± 120	<82	-	-	<77	<53	-	<8
	80-293	T-4	3/12/80	32.0	1	74.7	165 ± 5	-	30 ± 15	-	-	<50	<98	-	<5
	80-299	T-5	3/12/80	19.1	1	87.2	24 ± 7	-	<80	-	-	<55	<70	-	<5

Table 20. Radionuclide results for alder samples collected near Point Lepreau.

Sample	Sample No.	Type	Collection		Sample		% Ash	Cs-137 (Bq/kg)	Other Radionuclides (Bq/kg)
			Site	Date	Weight (g)	Geometry			
Alder (*Alnus rugosa*)	80-279a	branch (ash)	T-1	1/12/80	9.1 (ash)	4	0.98	3.18 ± 0.07	
	80-279b	buds	T-1	1/12/80	13.4	4	-	26.7 ± 3.3	
	80-283a	branch	T-2	3/12/80	66.6	2	-	5.5 ± 1.7	Ce-144 (4.6 ± 1.0)
	80-283a	branch (ash)	T-2	3/12/80	10.7 (ash)	4	1.0	4.4 ± 0.2	
	80-289a	branch (ash)	T-3	1/12/80					
	80-302a	branch (ash)	T-5	3/12/80					
	80-283b	buds	T-2	3/12/80	34.9	3	-	15.7 ± 1.8	
	90-289b	buds	T-3	1/12/80					
	80-295b	buds	T-4	3/12/80					
	80-302b	buds	T-5	3/12/80					
	80-295a	branch	T-4	3/12/80	20.9	3	-	3.2 ± 1.7	
	80-302a	branch	T-5	3/12/80	16.3	3	-	4.5 ± 2.5	

Table 21. Radionuclide results for blueberry leaves collected near Point Lepreau.

Sample No.	Collection Site	Collection Date	Sample Weight	Geometry	%H_2O	Cs-137 (Bq/kg)	Be-7 (Bq/kg)	Ce-144 (Bq/kg)	Ce-141 (Bq/kg)	Ru-103 (Bq/kg)	Ru-106 (Bq/kg)	Zr-95 (Bq/kg)	Nb-95 (Bq/kg)	Mn-54 (Bq/kg)
81-4660	T-4	14/7/81	71.9	1	8.2	253 ± 7	100 ± 27	80 ± 23	7 ± 3	<8	<30	13 ± 6	41 ± 13	<3
81-4661	T-3	14/7/81	46.2	1	7.6	103 ± 7	97 ± 40	88 ± 37	<28	<15	<48	10 ± 6	38 ± 24	<5
81-4662	T-2	14/7/81	50.2	1	5.7	56 ± 5	72 ± 28	89 ± 28	<17	15 ± 6	<38	23 ± 6	56 ± 20	4 ± 2
81-4663	T-1	14/7/81	35.9	1	5.1	58 ± 7	102 ± 47	205 ± 50	<35	<19	<67	42 ± 12	70 ± 48	5 ± 3
81-4664	T-7	14/7/81	43.8	1	5.4	15 ± 3	97 ± 45	77 ± 22	<16	<9	<20	35 ± 7	46 ± 19	5 ± 2
81-4666	T-10	14/7/81	26.0	1	4.8	83 ± 8	147 ± 47	90 ± 30	<50	<23	<80	83 ± 33		5 ± 3
81-4667	T-12	14/7/81	30.0	1	5.3	102 ± 5	160 ± 80	110 ± 35	<28	<14	<45	50 ± 12	38 ± 36	3 ± 2

(Liden and Gustafsson, 1967). Lichens are long-lived and have persistent above-ground parts, factors which further enhance their ability to accumulate fallout radioactivity (Gorham, 1959). In studies comparing radionuclide seed plants to lichen growing in the same area, lichens contained roughly two to ten-fold higher levels of radionuclide concentrations than the seed plants.

Lichens are particularly prominent in the food chain transport of radioactivity in arctic regions. These regions support a large population of reindeer and caribou which feed on lichen and which in turn provide an important food source for many humans living in these regions. High concentrations of Cs-137 were measured in a Alaska lichens in 1959, and high levels were detected in caribou in 1960 (Hanson, 1966). By 1962, Hanson had measured high levels of Cs-137 in Alaskan Eskimos which were comparable to those of the reindeer-breeding Lapps from northern Scandinavia, the latter due to the lichen-reindeer-man food chain. The importance of lichens in the passage of radionuclides to reindeer and caribou is promoted by the generally low mineral and nutrient content of lichen. Alaskan lichens contain only 10% to 50% of the potassium concentrations of higher plants such as willow and birch. The reduced potassium intake which occurs when organisms feed on lichen compared to other forms of vegetation tends to increase Cs-137 retention (Holleman et al., 1971). Because of their food link with lichens, caribou and reindeer contain up to tenfold higher concentrations of Sr-90 and Cs-137 compared to other large herbivores, such as moose, which occupy many of the same regions but which have different food preferences.

The remarkable ability of lichens to assimilate radioactivity is illustrated by the high radionuclide concentration levels in samples collected from Point Lepreau in 1980 and 1981. The much higher levels of fission products in samples collected in 1981 (compared to the 1980 samples) indicate that the source of these fission products is the October, 1980 Chinese nuclear weapons test. Hence, radionuclides such as Zr-95 and Mn-54 were easily detectable in samples collected in July, 1981 but were consistently below the detection limit during the summer of 1980. Note, however, that the Cs-137 activity in lichen does not vary significantly between 1980 and 1981. This is due to the fact that lichens are fairly long-lived and have been acting as a sink for Cs-137 for several decades. The Chinese test in 1980 contributed little additional Cs-137 compared to inputs recorded by the lichen for the 1960's and 1970's. Despite the considerable variability in Cs-137 activities in lichen samples collected from different locations, the average Cs-137 activities in samples collected in 1980 and 1981 are in remarkably good agreement (Figure 45).

Radionuclide concentrations vary markedly between different lichen samples collected at different locations. The actual radioactivity content of each sample reflects the age and size of the lichen, its efficiency at accumulating fallout radionuclides, the amount of precipitation at each location, drainage patterns, etc. The short-term efficiency of each lichen sample as an integrator, or sampling device for fallout radionuclides may be estimated from its concentration of Be-7, a naturally-produced radionuclide. The atmospheric flux of Be-7 is comparatively uniform, so lichen samples high in Be-7 should also have high concentrations of other short-lived fallout radionuclides. The overall efficiency of the lichens as detection instruments for Be-7 may be deduced from its mean average air concentration of 3.0 mBq/m^3 in air monitoring station samples (Table 19)

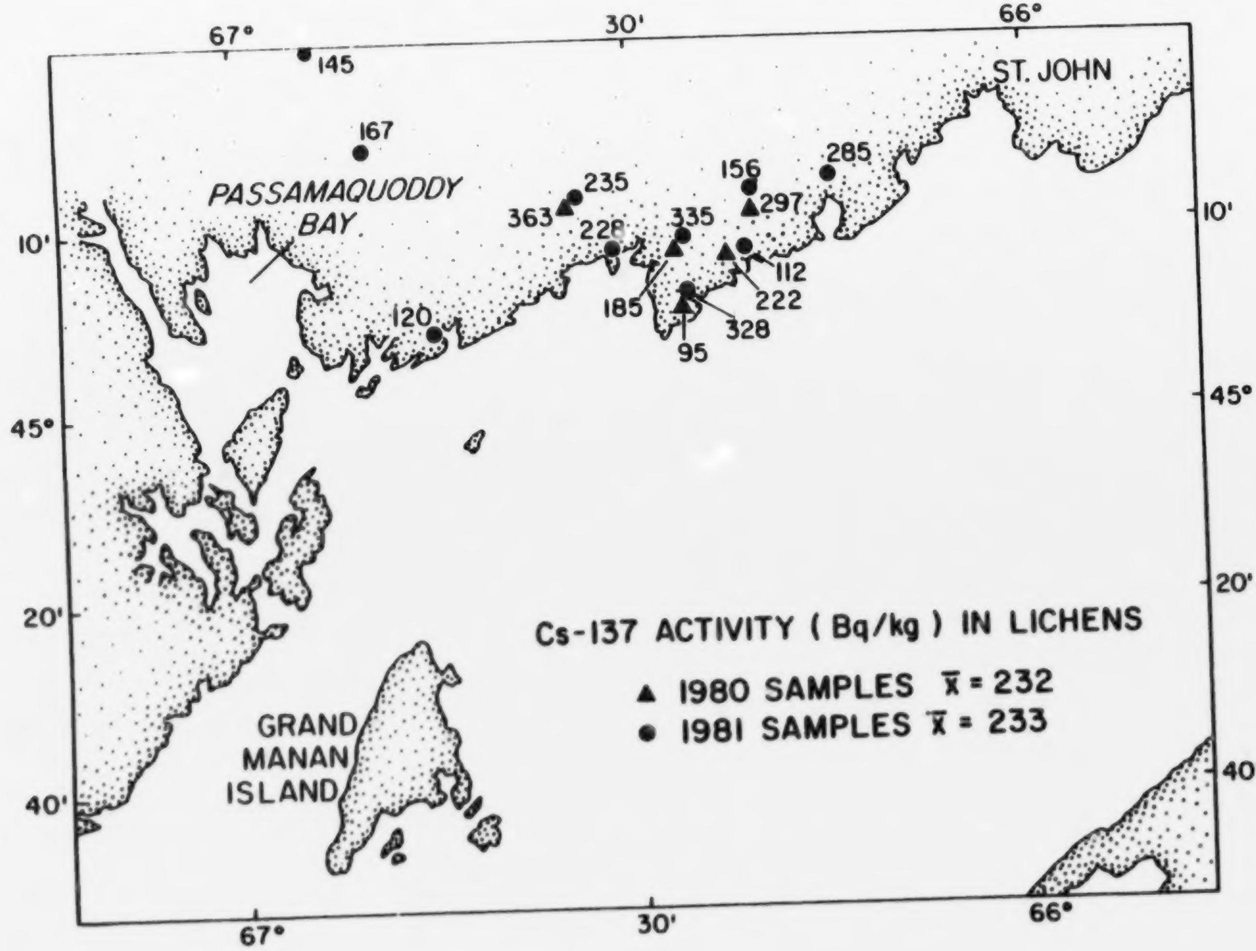


Figure 45. Distribution of lichen samples and their Cs-137 activity levels for 1980 and 1981 sample collection periods. Mean Cs-137 activity levels ($\bar{x}$) are almost identical for each year.

collected during July, 1981 and its mean concentration of 216 Bq/kg in lichen samples collected during the same period. From the ratio, Be-7 lichen activity/Be-7 air concentration = 72, it may be inferred that each gram of lichen samples approximately 72 m^3 of air during a period of several months. Hence, 10 grams of lichen sample approximately the same air volume (720 m^3) as one air monitoring station, although the actual sampling period is not as well defined as in the case of the air monitoring stations.

Most of the fallout radionuclides in the suite of lichen samples exhibit a linear relationship with respect to Be-7 with a correlation coefficient close to 0.9 (Figures 46 and 47) and a regression curve that goes near or through the origin. The wide range of Be-7 activities indicates that different lichen samples may have collection efficiences which differ by almost an order of magnitude. However, the good linear correlation between Be-7 and the shorter lived radionuclides, Zr-95, Ru-103 and Ce-141 indicates that over sampling periods of the order of a few months, the collection efficiency of each lichen sample remains constant for the entire range of fallout radionuclides. This is an important result because it means that the Be-7 content of a lichen sample can be used to calibrate the collection efficiency of this sample for the uptake of other airborne radionuclides. If, for example, a radionuclide suspected to be of reactor origin is detected in a lichen sample, then the Be-7 content of that sample provides a collection efficiency measurement which can then be used to estimate the air concentration of the reactor-derived radionuclide. What is somewhat unusual about the lichen results in Figure 47 is the good linear correlation between Be-7 and the longer-lived radionuclide, Cs-137. This correlation indicates that the collection efficiency of these lichen

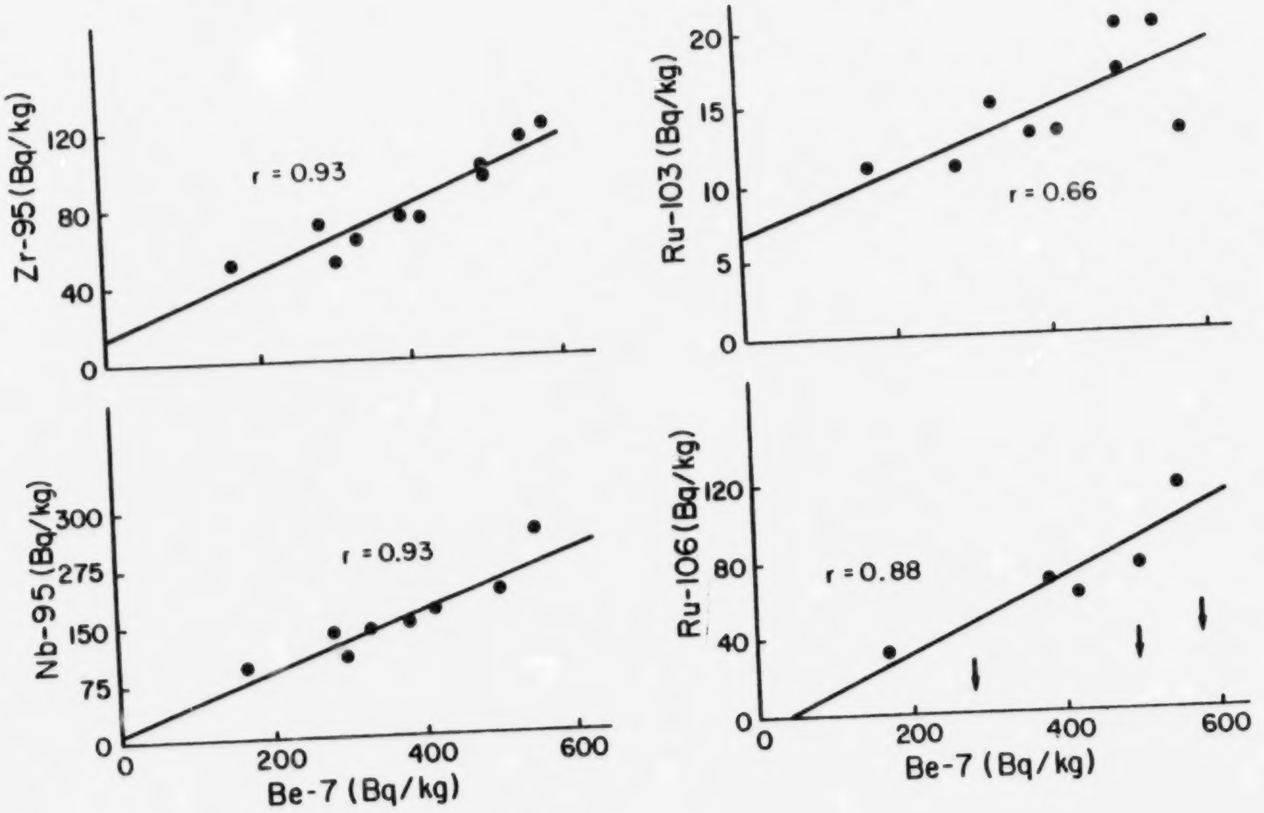


Figure 46. Relationship between fallout radionuclide levels in lichen and their Be-7 content. Arrows indicate values below the detection limit. Least squares, linear fit having high correlation coefficient (r) in most cases indicates that Be-7 is good calibration radionuclide for estimating collection efficiency of lichen for broad range of radionuclides.

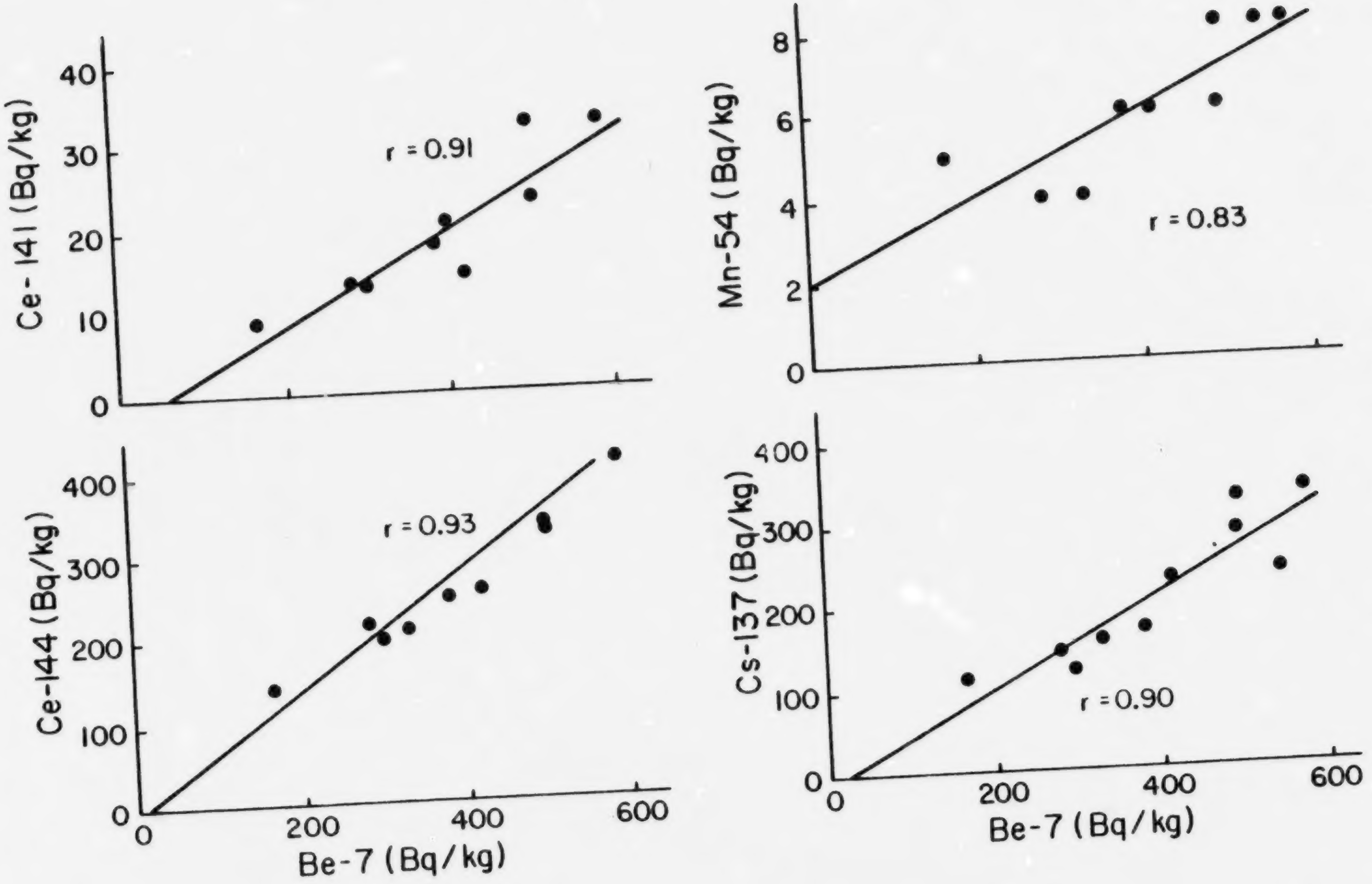


Figure 47. Relationship between fallout radionuclide levels in lichen and their Be-7 content. Linear correlation curve is close to orgin for most radionuclides, although small offset is observed for Mn-54.

samples has not changed appreciably between the 1960's and 1970's (when most of the Cs-137 was presumably accumulated) and the present. These results suggest that lichen is an equally good monitoring instrument for the longer-lived radionuclides, provided reactor-source inputs can be distinguished from baseline levels.

Anderson et al. (1957) reported that the Cs-137 uptake by plants growing near the Oak Ridge nuclear laboratories was inversely proportional to the exchangeable potassium in the soil. In laboratory experiments, a twofold increase in the soil potassium resulted in a decrease in the Cs-137/K ratio in alfalfa by a factor of 9 (Fowler and Christenson, 1959). In view of the many observations of this competition between potassium and Cs-137 for assimilation in natural systems (Davis 1963), it is perhaps not surprising that the Cs-137 content of lichen samples collected in the Point Lepreau region is inversely proportional ($r = -0.70$; $n = 15$) to the K-40 content of the samples (Figure 48). Note that this relationship holds for samples collected both before and after the 1980 Chinese weapons test. Hence, it appears that the two principal factors governing the distribution of Cs-137 in lichen samples are (1) the retention efficiency of the lichen for fallout radionuclides, as indicated by the Be-7 activity, and (2) the quantity of exchangeable potassium competing with Cs-137 for uptake.

In section 4.2.1 of this report, the fission product ratios measured in atmospheric particulate samples were compared to the predicted ratios determined from the theoretical yields of the nuclear weapons. It is instructive to compare both sets of results with the fission product ratios measured in lichen samples. For example, the mean Zr-95/Nb-95 ratio for the 11 lichen samples collected in July, 1981 is 0.51, which is in almost perfect agreement with the predicted value of 0.50 (Figure 27). In

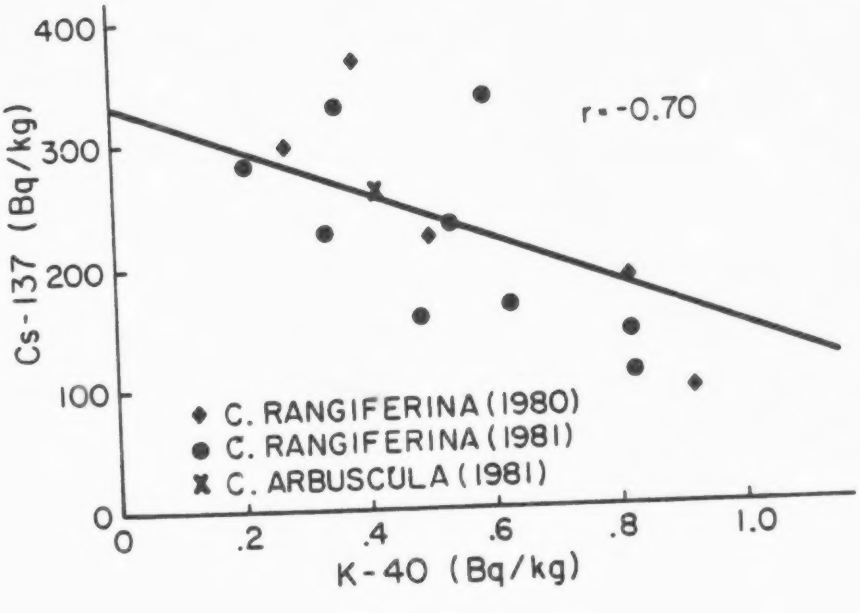


Figure 48. Cs-137 activity levels in different types of lichen as a function of
K-40 content. Inverse, linear relationship has correlation coefficient
of -0.70.

fact, the lichen samples provide a more accurate and more precise measurement of this ratio than do the air particle measurements. Similarly, the mean ratio of Ru-103/Zr-95 of 0.18 in lichen samples is only slightly lower than both the mean ratio measured in atmospheric particulates and the predicted ratio of 0.25. Best agreement would be expected between the lichen and theoretical results for short-lived isotopes because there will be negligible residual background of these isotopes from previous nuclear weapons tests.

The mean Ru-103/Ru-106 ratio in lichen of 0.24 is again only slightly less than the predicted ratio of 0.33 (Figure 31). The decreased ratio in lichen may be due in part to residual Ru-106 deposited prior to the Chinese weapons test in 1980. Unfortunately, it is not possible to determine the pre-1980 background Ru-106 levels on lichen with a great degree of accuracy because the 1980 samples were only analysed several months subsequent to their collection and the detection limits for many of the shorter-lived radionuclides are quite high. The mean Ru-103/Ce-144 and Zr-95/Ce-144 ratios in lichen of 0.054 and 0.29, respectively are considerably lower than both the atmospheric particulate measurements of these ratios and the predicted ratios of 0.125 and 0.48, respectively, (Figures 33 and 34). If there has been no differential removal of these radionuclides from lichen surface, which appears to be a valid assumption based upon the previous examples, then the above results indicate that approximately half of the Ce-144 on the lichen samples has been derived from previous (pre-1980) nuclear weapons tests. By far the poorest agreement between predicted and measured isotopic ratios occurs for cases in which Cs-137 is considered. The mean Ce-144/Cs-137 and Zr-95/Cs-137 ratios of 1.4 and 0.39, are much smaller than the air particulate ratios and the predicted ratios of 17.3 and 8.0, respectively. Again, these results could be explained if approximately 90% - 95% of the Cs-137 retained on the lichen had been deposited prior to the 1980 nuclear weapons test, a hypothesis which is plausible in view of the long lifetime of the lichen and the much greater Cs-137 fallout inputs which occurred during the 1950's and 1960's.

Lichen plants are excellent indicators and integrators of fallout radionuclides and in some instances they perform quite reliably the function of the atmospheric particulate samplers of the air monitoring stations. They provide a reasonable measurement of the total fallout input and of isotopic ratios in fallout and they may be calibrated by using the Be-7 content as a collection efficiency measurement. Their chief drawback as air samplers is that they integrate over the entire lifetime of the plant and do not give a instantaneous measurement of the air concentration of a radionuclide. On the other hand, as long term indicators of fallout radioactivity they are extremely useful and lichen monitoring is, of course, very cost effective.

Moss, Alder and Blueberry Leaves

Sphagnum (moss) generally grows as a compact mat which, when saturated with water, facilitates foliar sorption of radionuclide ions from solution. Measurements of the fallout interception efficiency of Sphagnum bogs in the Colorado Front Range in 1962 and 1963 indicated that essentially all the Cs-137 and Sr-90 deposited was accounted for as an increment in the Sphagnum strate (Osburn, 1969). Cesium-137 concentrations of the order of 200 Bq/kg were measured in (Svaboda and Taylor, 1979) Sphagnum samples collected in Vermont, Ontario and the Northwest Territories during the early 1970's. The principal differences in moss and lichen as indicator species for radionuclides is that moss tends to assimilate radionuclides by sorption from solution while lichen tends to be a dry air particulate collector. The somewhat greater efficiency of lichen, compared to moss, as a collector of Cs-137, is evidenced by the higher levels of Cs-137 measured in the lichen compared to those measured in moss for samples collected in 1980 of POint Lepreau (Table 19). However, this comparison is problematic, because atmospheric Cs-137 levels are always greater during the spring and summer months (when the lichen samples were collected) compared to the winter months when the moss samples were collected. Further, each plant species is in contact with a different environmental compartment, so comparisons between the two are somewhat arbitrary. Unfortunately, no moss samples were collected during the summer of 1981, and uptake of the short-lived fission products from the Chinese weapons test was not measured in moss. Mosses are useful indicator species, providing a good measurement of radioactivity levels circulated through the terrestrial hydrological system, and reliance will be placed on these species in future monitoring at Point Lepreau.

Alnus rugosa (speckled alder) commonly occurs in thickets in wet soil along streams, lakes and swamps and its occurrence in drainage areas makes it a good indicator for water-borne radionuclides. Alder is not a primary browse species, but is is located at the base of the food web and is utilized by insects which are themselves consumed by amphibians, birds, mammals and reptiles. Previous experiments concerning the detection of radionuclides in deciduous trees include a study by Olson et al. (1960) of Cs-134 movement in a white oak (Quercus albai) forest. Subsequent to injection of tracer Cs-134 into oak tree bore holes this radionuclide was observed to move rapidly into tree tops and within 30 days after inoculation, 50% to 70% of the tracer cesium had moved into the leaves and buds. The Cs-137 measurements on alder samples from Point Lepreau (Table 20) also exhibit substantially higher levels in the buds compared to the branches. Cesium-137 is easily detected in all components of the alder, and this appears to be a good indicator plant for wet, poorly drained habitats.

Cesium-137 concentrations measured in 12 plant species collected within a single drainage system in the northern Colorado Front Range in 1963 and 1964 generally varied between 30 Bq/kg and 300 Bq/kg (dry) (Remmenga and Whicker, 1967). These samples were collected at a time (1963-64) when fallout deposition of Cs-137 was maximized. Considerable variation between species growing in the same sampling sites was noted. For example, blueberry contained about four to seven times the Cs-137 activity of aspen from sapling sites less than 10 m apart. These differences can be ascribed to variations in morphology. The prostrate form and fine branching of blueberry facilitates a more intimate contact with the melting snow pack and direct transfer of radionuclides to the vegetation surfaces. Blueberry plants collected at Point Lepreau are similarly observed to be good indicators of various radionuclides. They are particularly efficient at collecting Cs-137 and are characterized by levels of this radionuclide substantially greater than those measured in alder collected from the same sampling site (Table 21). However, as in the case of horse tail, Cs-137 levels in blueberry leaves are quite variable and cover a range of from 15 Bq/kg to 253 Bq/kg. There is no indication from other indicator species (eg. lichen) that these the sampling sites for the blueberry samples noted above have been exposed to significantly different atmospheric Cs-137 levels, so it may be assumed that differences in the blueberry radionuclide levels are mainly due to differences in the plants themselves.

Uptake of radioactivity in blueberry occurs by transport through the root system in addition to direct uptake in the leaves and stems. The more complex physico-chemical processes governing radionuclide uptake in blueberry is evidenced by the absense of a simple linear relationship between the Cs-137 and Be-7 activities, as occurred in the case of lichen (Figure 49). Clearly, the Cs-137 content of blueberry is not simply a linear function of the quantity of atmospheric particulates (as characterized by the Be-7 activity) to which the plants have been exposed. In this regard, it may be noted that marsh grass, an emergent aquatic plant which also assimilates Cs-137 through its roots and through its leaf system, exhibits a variability in its Cs-137 levels which is similar to that exhibited by blueberry leaves (Figure 49). The correlation coefficient for a linear correlation between Cs-137 and Be-7 is 0.073 (n = 7) for blueberry leaves and -0.207 (n = 5) for marsh grass compared to a value of 0.90 (n = 10) for lichen. Blueberry leaves are effective in accumulating fallout raidonuclides, as evidenced by the detection of the short-lived fission products (Ce-141, Ce-144, Zr-95 etc.) derived from the 1980 Chinese weapons test. In general, fallout radionuclide levels in blueberry leaves are a little less than half of the radionuclide levels measured in lichen collected the same time and locations. However, the mean isotopic ratios in most cases are within 20% to 30% of those measured in lichen samples and many of the comments made regarding the lichen results also apply to blueberry leaves. In general, blueberry leaves are good indicators for fallout radionuclides, but the data is probably a little less reliable and more difficult to interpret compared to the lichen data.

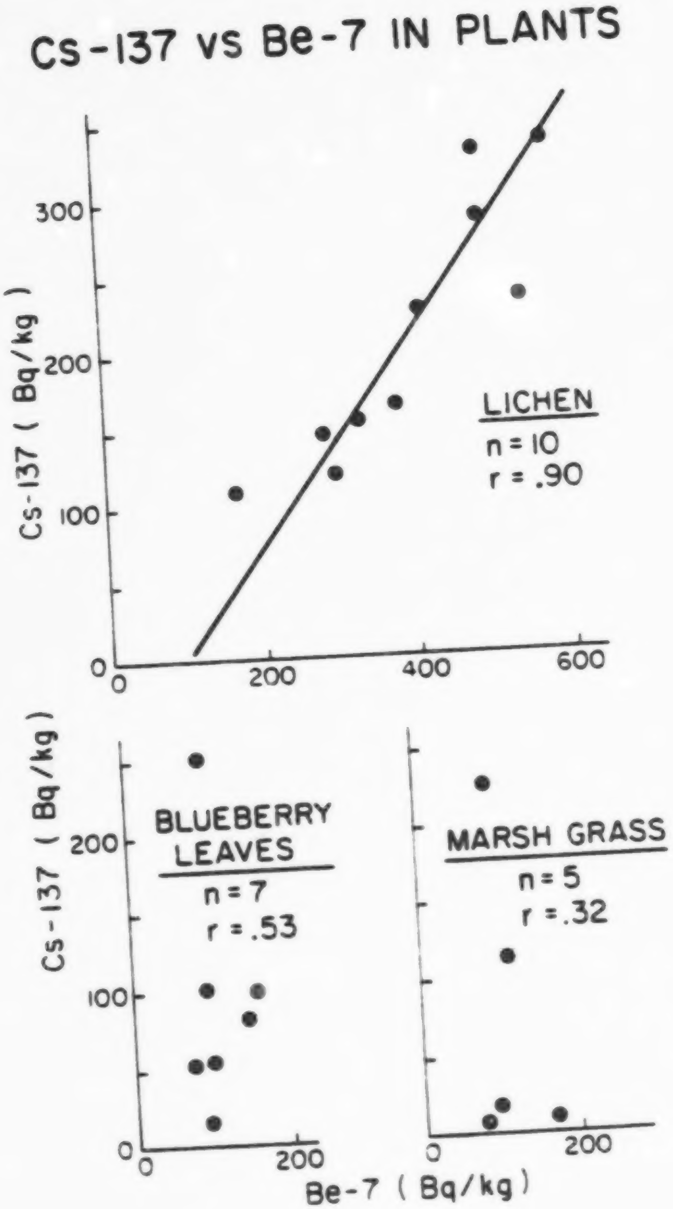


Figure 49. Relationship between Cs-137 activity and Be-7 activity in lichen, blueberry leaves and marsh grass. Absense of linear relationship in latter two plants indicates that processes other than simple physical sorption may be significant in radionuclide uptake.

6.2.3 Terrestrial Vertebrates

The large variations in radionuclide concentrations measured among different species of vegetation points to the importance of food habits in the accumulation of radionuclides by herbivores. Many of the studies concerning radioactivity uptake by herbivores has focused on deer as these are important agents for radionuclide transport to carnivores and a potential pathway to humans (Whicker et al., 1967; Hanson et al., 1963; Longhurst et al., 1967). Measurements of Cs-137 in mule deer collected in Colorado between 1962 and 1965 gave values ranging from 7.4 to 150 Bq/kg (Whicker et al., 1965) which followed a distinct seasonal pattern. Deer flesh in Colorado in the mid-1960s contained five to ten times the concentrations of Cs-137 as beef or pork. It was also noted that the consumption of deer liver could lead to appreciable uptake of Ce-144, Cs-137, Mn-54 and Ru-106, all of which are concentrated in this organ. The transfer of Cs-137 from mule deer to a principal predator, the mountain lion (*Felis concolor*) has received some study and it has been noted that a three fold increase in the Cs-137 concentration accompanies this transfer (Pendleton et al., 1964).

The three principal herbivores monitored for radioactivity at Point Lepreau are the muskrat (*Ondatra zibenthicus*), varying hare (*Lepus americanus*) and meadow vole (*Microtus pennsylvanicus*). Deer tissues will also be examined in the future when they become available. The muskrat is an amphibious rodent that spends most of its time in the water. During the summer it feeds upon emergent vegetation (cattail, horse tail, water lily, etc.) and during the winter it feeds upon submerged vegetation (water lily tuber, pondweed etc.). The highest Cs-137 levels in muskrats collected at Point Lepreau were measured in the muscle tissue (Table 18). This result is in agreement with previous work which has indicated that voluntary muscle of terrestrial mammels is generally the dominant accumulator of cesium, both on total body and per gram tissue basis. Ballou and Thompson (1958) found that 57% of the total dose of Cs-137 administered to rats accumulated in muscle and noted that about 80% accumulates in the muscles of cattle, sheep and pigs. Nelson et al. (1961) conducted a study in which Cs-137 uptake was traced throughout the tissues of mice. The measured distributions 36 days after the administration of the tracer were roughly consistent with the results for muskrat given in Table 18 with the highest levels measured in the muscle and the lowest measured in bone. It is interesting to note that the Cs-137 concentration factors for muskrat tissues are approximately the same order of magnitude as those for the aquatic plants which they are known to consume (Table 17). Hence, there is no obvious concentration or discrimination effect in Cs-137 transport between plants and muskrats. However, the Co-60 concentration in brain tissue (23 Bq/kg) is much higher than levels measured either in vegetation or other tissues of the muskrat. This organ exhibits a marked concentrating effect for cobalt.

The varying hare is commonly distributed throughout the forested regions of Canada and lives on a wide variety of green grasses, blueberry leaves, clovers and horsetails during the summer and buds, twigs and evergreen leaves during the winter. The Cs-137 levels in the hare were greatest in the muscle tissue, approximately twice as great as levels measured in muskrats. The slightly higher levels of Cs-137 in hares compared to muskrats may reflect the slightly greater concentrations of this radionuclide in terrestrial plants compared to aquatic plants. The meadow vole is the ubiquitous 'field mouse' which feeds primarily on grasses and sedges lives in wet, grassland habitats and feeds primarily on grasses and sedges.

sedges. In spring and summer they feed on growing shoots and in winter they eat seeds, roots and the basal green portions of the grass beneath the snow. Both Cs-137 and Co-60 were easily detected in the meadow vole (muscle + liver + intestines) at levels similar to those measured in both aquatic and terrestrial vegetation.

Cesium-137 concentrations in aquatic and terrestrial plants and in the muscles of herbivores, which tend to feed on these plants, are of the same order of magnitude (Figure 50). Hence, there is very little change in Cs-137 activities during transport through this link in the food chain. However, there is a great difference in Cs-137 levels between the different tissues of terrestrial mammals and the same tissues in a marine mammal (harbour seal; Figure 51). Cesium-137 levels in terrestrial mammals are approximately two orders of magnitude greater than the levels in harbour seals. This is consistent with the previous observation that Cs-137 levels in aquatic plants and fish are also two orders of magnitude greater than levels in marine plants and fish. Clearly, biological uptake of Cs in high ionic strength, marine systems is far less efficient compared to uptake in terrestrial systems.

6.3 Birds

Birds, like marine vertebrates, are not particularly suitable as indicator species for radionuclide uptake because of their mobility over a comparatively wide range. They do, however, occupy an important position in the food web, particularly in the Bay of Fundy, which is a critical location for foraging by a large number of migratory shorebirds before migration to southern regions. Uptake of radionuclides occurs by sorption

Cs-137 IN TERRESTRIAL SAMPLES

SAMPLE	ACTIVITY Cs-137 (Bq/kg) RANGE	AVERAGE
MARSH GRASS	5-233	62
LICHEN	95-363	233
SPHAGNUM	58-165	99
BLUEBERRY LEAVES	15-253	96
MUSKRAT MUSCLE		183
HARE MUSCLE		410
MEADOW VOLE		37
SOIL(A HORIZON)	10-80	

Figure 50. Cs-137 activities in different terrestrial phases.

Cs-137 ACTIVITY (Bq/kg (DRY)) IN MAMMALS

* INCLUDES KIDNEY	TERRESTIAL			MARINE
	MUSKRAT (3)	HARE (1)	MEADOW VOLE (1)	HARBOUR SEAL (3)
MUSCLE	173	410	–	1.5
LIVER	82	283	–	1.6
BRAIN	77	80*	–	1.5
ALL			37	
% MOISTURE	71	79	84	70

Figure 51. Cs-137 activities in different tissues of terrestrial and marine mammels.

of radionuclides in food or other ingested materials or by direct adsorption of radionuclides from air or from solution.

Twenty-one artificial radionuclides have been measured by Markham and Halford (1982) in mourning doves in the Idaho National Engineering Laboratory (INEL) Site in southwest Idaho. Highest activities found were for the radionuclides Cs-137, Cr-51, Co-60, I-131 and Ce-144. Activities of Cs-137 of up to 6300 Bq/kg dry weight in muscle and up to 16,000 Bq/kg in gastrointestinal tracts were measured. Background levels of Cs-137 were generally below 22 Bq/kg. Willard (1960) observed that Cs-137 levels in bird muscle increased dramatically in winter as a result of ingestion by birds of contaminated soil while foraging for food.

The birds collected for monitoring purposes at various sites near Point Lepreau (Figure 52) are, (marine environment); (1) Phalocrocorax carbo (great cormorant), Somateria mollissima (eider duck), Larus argentatus (herring gull) and Calidris maritma (purple sandpiper); (terrestrial environment); Sturnus vulgaris (starling) and Melospiza melodia (song sparrow). Cormorants are common throughout the Point Lepreau area, the great cormorant being a transient, winter resident.Their principle prey is fish, particularly sculpin and herring, with the balance of their diet being amphipods, seaweed, shrimp and crabs. The common eider is a year round resident of the Bay of Fundy, feeding exclusively on tidal and shallow, subtidal plants and animals. Its diet consists of Mytilis edulis, sea urchins, Thais lapillus, and debri, determined from gut content analyses (Marine Research Associates Ltd., 1979). The herring gull is also a common resident of the Bay of Fundy with population estimates of greater than 100,000 (Squires, 1976). Herring gulls have a mixed diet of marine and terrestrial materials, depending on the availability of food. They are

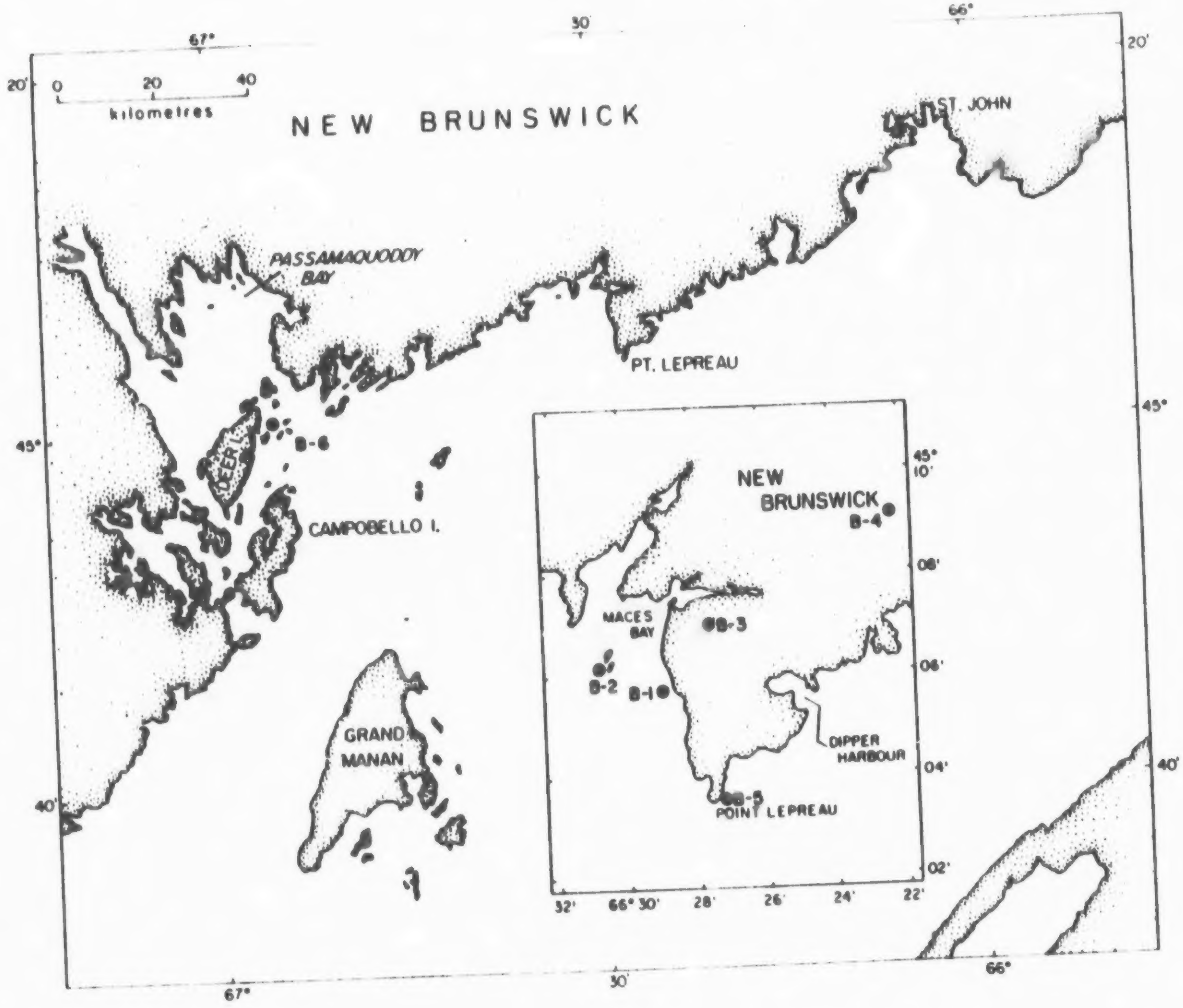


Figure 52. Sampling sites for birds collected in vicinity of Point Lepreau.

active scavengers feeding on pelagic fish and krill and to a lesser extent on intertidal and shallow subtidal organisms and on any available food material on the shore or land. The purple sandpiper is generally a winter inhabitant, feeding on intertidal littoral grazers, detritus feeders and filter feeders. The starling is a year round inhabitant whose diet consists mainly of insects (including grubs, caterpillers and adults) as well as fruit, berries, grains, seeds and garbage. The song sparrow is also a common year round resident, feeding mainly on weed seeds and insects.

The results for the birds, which were separated into muscle, liver, bone and gut parts, are given in Tables 22-24. Cs-137 and Co-60 were the only artificial radionuclides measured, with the exception of the herring gull liver sample which contained Mn-54. Highest activities were measured in starlings (up to 18 Bq/kg dry) and song sparrows (average Cs-137 activity of 3-4 Bq/kg, Table 22) with lowest activities measured in herring gulls, cormorants, eider ducks and sandpipers (Table 23 and 24). The higher levels found in the terrestrial birds reflect the higher levels of radioactivity present in their food (terrestrial plants and insects) compared to levels found in the food of the marine birds. As noted previously in this report levels of radioactivity in terrestrial plants are an order of magnitude greater than those in marine organisms. For example, Cs-137 activities measured in alder buds (20 Bq/kg), typical of terrestrial bird food, are about 10 times the activity of herring muscle (1.1 Bq/kg), typical of marine food. The slightly higher levels (up to 1.8 Bq/kg) found in the herring gull compared to other marine birds probably reflects its mixed marine-terrestrial diet. Levels of Cs-137 and Co-60 are similar and are highest in muscle and liver. For the starling, both Cs-137

Table 22. Radionuclide results for terrestrial birds collected near Point Lepreau.

Species	Sample No.	Organ	Collection		Sample		%H_2O	Cs-137 (Bq/kg)	Co-60 (Bq/kg)
			Site	Date	Weight (g)	Geometry			
Starling	80-269a	muscle	B-4	16/2/80			74.6		
(Sturnus	80-270a	muscle	B-4	16/2/80	13.3	3	70.8	18 ± 10	18 ± 12
vulgaris)	80-271a	muscle	B-4	16/2/80			72.3		
	80-269b	liver	B-4	16/2/80			76.4		
	80-270b	liver	B-4	16/2/80	2.02	5	79.3	8.7 ± 4.3	10 ± 7
	80-271b	liver	B-4	16/2/80			77.4		
	80-338c	bone	B-4	16/2/80			63.6		
	80-339c	bone	B-4	16/2/80	11.4	4	57.8	<2.8	<4.1
	80-340c	bone	B-4	16/2/80			60.3		
	80-338d	gut	B-4	16/2/80			71.9		
	80-339d	gut	B-4	16/2/80	11.4	4	45.4	<4.0	2.3 ± 1.6
	80-340d	gut	B-4	16/2/80			63.8		
Song Sparrow	80-344	total	B-6	12/2/80	10.8	3	55.2	4.3 ± 2.0	3 ± 2
(Melospiza	80-345	total	B-6	12/2/80	12.0	3	52.5	3.2 ± 1.7	<7.0
melodia)	80-346	total	B-6	12/2/80	12.5	3	55.2	2.8 ± 1.8	<8.2

Table 23. Radionuclide results for marine birds collected near Point Lepreau.

Species	Sample No.	Organ	Collection		Sample		%H_2O	Cs-137	Co-60
			Site	Date	Weight (g)	Geometry		(Bq/kg)	(Bq/kg)
Eider (**Somateria mollissima**)	80-260a	muscle	B-1	9/2/80	43.65	2	71.2	<2.3	1.7 ± 1.2
	80-261a	muscle	B-1	9/2/80	43.62	2	72.0	<1.2	1.3 ± 0.8
	80-262a	muscle	B-1	9/2/80	45.79	2	69.9	<2.5	<1.7
	80-260b	liver	B-1	9/2/80	18.0	4	62.7	<3.3	<3.3
	80-261b	liver	B-1	9/2/80	15.6	4	61.7	<5.0	<6.7
	80-262b	liver	B-1	9/2/80	13.0	4	60.8	<5.3	<8.0
	80-329c	bone	S-1	9/2/80	21.4	4	42.7	<3.2	<4.8
	80-330c	bone	B-1	9/2/80	36.3	3	34.5	<2.0	<2.7
	80-331c	bone	B-1	9/2/80			39.4		
	80-329d	gut	B-1	9/2/80	96.7	2	70.7	<1.1	0.8 ± 0.5
	80-330d	gut	B-1	9/2/80	76.5	2	71.9	<1.6	<2.2
	80-331d	gut	B-1	9/2/80	70.1	2	70.0	<1.7	<2.3
Cormorant (**Phalacrocorax carbo**)	80-263a	muscle	B-2	9/2/80	56.3	2	68.8	<1.3	1.5 ± 1.0
	80-264a	muscle	B-2	9/2/80	48.2	2	70.5	1.5 ± 0.7	<1.8
	80-265a	muscle	B-1	9/2/80	58.2	2	68.9	<1.3	<2.7
	80-263b	liver	B-2	9/2/80	29.1	3	67.8	1.7 ± 1.0	<3.5
	80-264a	liver	B-2	9/2/80	22.6	3	69.4	<2.8	1.8 ± 1.3
	80-265b	liver	B-1	9/2/80	21.8	3	68.6	<3.3	<3.2
	80-332c	bone	B-2	9/2/80	25.0	4	38.4	<2.8	<4.2
	80-334c	bone	B-1	9/2/80			40.1		
	80-333c	bone	B-2	9/2/80	13.3	4	40.4	<2.5	<7.8
	80-332d	gut	B-2	9/2/80	159.9	1	51.9	0.7 ± 0.3	<1.0
	80-333d	gut	B-2	9/2/80	126.0	1	54.0	0.4 ± 0.3	<1.3
	80-334d	gut	B-1	9/2/80	147.1	1	67.4	0.4 ± 0.2	0.8 ± 0.5

Table 24. Radionuclide results for marine birds collected near Point Lepreau.

Species	Sample No.	Organ	Collection Site	Collection Date	Sample Weight (g)	Sample Geometry	%H_2O	Cs-137 (Bq/kg)	Co-60 (Bq/kg)	Other Radionuclides (Bq/kg)
Herring Gull (*Larus argentatus*)	80-266a	muscle	B-3	16/2/80	21.8	3	65.3	1.8 ± 1.0	<1.5	
	80-267a	muscle	B-3	16/2/80	22.6	3	66.8	1.5 ± 0.8	1.3 ± 0.4	Mn-54 4.0 ± 1.5
	80-268a	muscle	B-3	16/2/80	26.1	3	64.5	1.7 ± 0.7	1.3 ± 0.7	
	80-266b	liver	B-3	16/2/80	28.4	3	59.3	<2.3	<3.2	
	80-267b	liver	B-3	16/2/80			58.2			
	80-268b	liver	B-3	16/2/80			52.5			
	80-335c	bone	B-3	16/2/80	18.0	4	40.7	<3.0	<8.2	
	80-337c	bone	B-3	16/2/80	12.2	4	31.8	<4.2	4 ± 2	
	80-336c	bone	B-3	16/2/80			43.7			
	80-335d	gut	B-3	16/2/80	18.4	3	72.1	<4.0	<5.0	
	80-336d	gut	B-3	16/2/80	26.6	3	70.4	1.0 ± 0.8	<3.7	
	80-337d	gut	B-3	16/2/80	18.5	3	68.8	<3.7	<3.5	
Purple Sand-piper (*Arquatella maritima*)	80-272a	muscle	B-5	9/2/80	11.4	4	75.0	<5.0	<8.2	
	80-273a	muscle	B-5	9/2/80			73.9			
	80-274a	muscle	B-5	9/2/80			73.2			
	80-272b	liver	B-5	9/2/80	2.8	5	77.3	<3.9	9 ± 5	
	80-273b	liver	B-5	9/2/80			78.8			
	80-274b	liver	B-5	9/2/80			76.0			
	80-341c	bone	B-5	9/2/80	5.0	5	64.3	<7	<10	
	81-342c	bone	B-5	9/2/80			62.3			
	80-343c	bone	B-5	9/2/80			63.3			
	80-341d	gut	B-5	9/2/80	11.3	5	66.7	<3.5	<5.2	
	80-342d	gut	B-5	9/2/80			66.1			
	80-343d	gut	B-5	9/2/80			63.9			

and Co-60 are twice as high in muscle as in liver, a result expected for Cs-137 (Davis, 1963) but surprising for Co-60, which is usually accumulated more effectively in the liver (Burton, 1975). The activities in the guts were found to be between 0.4 to 1.0 Bq/kg for Cs-137 and 0.8 Bq/kg for Co-60 in marine birds and 4.0 Bq/kg (Cs-137) and 2.3 Bq/kg (Co-60) for terrestrial birds, results which are accord with the differences in radionuclide levels in their respective foods. Activities measured in marine birds are similar to those measured in marine vertebrates, suggesting that they feed on similar foods and accumulate radionuclides in a manner similar to marine vertebrates. Activities measured in the terrestrial birds agree well with levels measured by Willard (1960) for birds in an uncontaminated environment. Activities are generally an order of magnitude lower than those in marine mammals and slightly lower than those in other terrestrial and aquatic vertebrates, as was also observed by Davis (1963). This probably reflects differences in habitat, feeding habits and food.

7.0 CONCLUSIONS

1. Results from the pre-operational component of the Point Lepreau environmental monitoring program have provided insight into the nature of the radionuclide transport pathways and mechanisms which will govern the distribution of radioactivity released from the Point Lepreau NGS. Baseline levels of key radionuclides have been measured in a broad range of environmental phases.

2. Results from the drifter return program have been used to identify potential transport pathways for radionuclides released into the ocean from the Point Lepreau NGS in both particulate and dissolved phases.

3. Water column concentrations of Cs-137, Sr-90 and tritium measured in both marine and freshwater systems near Point Lepreau are consistent with concentrations predicted on the basis of nuclear weapons fallout and the production rate of tritium in the atmosphere.

4. Sediment inventories of Pb-210, Cs-137 and plutonium radionuclides measured in the Bay of Fundy have been used to identify fine-grained depositional sinks which may become active sites for the future accumulation of reactor-released radionuclides. The distribution of radioactivity in these sediments is controlled by sedimentation, bioturbation and resuspension phenomena.

5. Radionuclide measurements performed on air particulate material collected in air monitoring stations located at Point Lepreau are in good agreement with synchronous measurements performed by other laboratories in the U.S. and Canada. Isotopic ratios (eg. Ce-141/Ce-144; Ru-103/Zr-95) measured for radionuclides produced during a 1980 Chinese nuclear weapons test have been used as a diagnostic to determine the source and timing of radionuclide inputs into the environment.

6. Baseline radionuclide concentrations and concentration factors for marine organisms collected near Point Lepreau are generally low and in agreement with literature results. Significant discrimination is observed in radionuclide levels between different tissues of individual organisms, but there are no apparent increases in radionuclide concentrations with increasing trophic level of the organism.

7. Baseline radionuclide levels are greater by several orders of magnitude in terrestrial and aquatic biological phases compared to marine biological phases. The lower levels in marine phases are due both to the presence of a large diluting seawater reservoir and the high ionic strength of seawater which acts to limit the sorption of radionuclides onto particle surfaces.

8. Lichen are particularly efficient accumulators of air particulate material and serve as a good indicator species for the time-integrated inventory of radioactivity in the atmosphere. Lichen are cost-effective air particulate samplers and the radioactivity determinations have an excellent signal to noise ratio.

9. The record of fallout radionuclide fluxes is preserved better in aquatic sediments than in more easily eroded soils, although details of the former record are obscured in many cases by sediment mixing due to biological activity.

10. The highest fallout radionuclide levels were measured in the organs of birds having a terrestrial diet while reduced levels were found in birds which tend to feed on marine organisms, a result which is consistent with the generally higher levels of radioactivity measured in terrestrial environmental phases.

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Appendix 1. Conversion factors for different radioactivity units.

1 Becquerel (Bq) = 2.7×10^{-11} Curies (Ci)

1 Bq = 27 pCi

1 Bq = 60 dpm

1 Bq/l = 8.3 Tritium units (TU)

1 Ci = 3.7×10^{10} Bq

1 mCi = 3.7×10^{7} Bq

1 pCi = 3.7×10^{-2} Bq

1 dpm = 0.0167 Bq

1 dpm = 2.2 pCi

1 TU = 0.12 Bq/l

1 TU = 7.2 dpm/l

1 TU = 3.2 pCi/l

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Nuclide	Half-life	Geometry 1	Geometry 2	Geometry 3	Geometry 4	Geometry 5
Ag-110 m	253 d	150	94	95	95	47
Ba-140	12.8 d	1700	1700	1000	1000	490
Be-7	53.6 d	1100	1200	690	670	370
Ce-141	32.4 d	280	330	170	140	110
Ce-144	284 d	780	900	460	400	200
Co-56	77 d	130	140	90	90	40
Co-57	270 d	110	130	63	54	35
Co-58	71 d	130	140	77	78	39
Co-60	5.26 y	120	120	65	69	35
Cr-51	28 d	1700	1800	1100	1001	570
Cs-134	2.07 y	110	120	69	69	33
Cs-137	30.2 y	107	100	62	60	31
Eu-152	12.5 y	660	690	410	410	200
Eu-155	1.81 y	300	340	160	160	69
Fe-59	44.6 d	340	330	190	200	89
I-131	8.06 d	1200	1400	780	750	410
La-140	40.2 h	510	510	340	360	190
Mn-54	313 d	110	110	67	65	31
Nb-95	35.1 d	180	180	110	110	54
Ru-103	39.6 d	150	160	93	92	46
Ru-106	1.0 y	900	940	540	550	250
Sb-124	60.2 d	18	20	11	11	6
Sb-125	2.7 y	240	260	150	147	73
Zn-65	244 d	290	260	140	150	73
Zr-95	64 d	240	240	150	150	70

Appendix 3. Detection limits (Bq/kg) for each radionuclide and geometry assumming a counting time of 10^5 sec for analyses performed one month subsequent to sample collection. Detection limit for specific sample is determined by choosing detection limit for a radionuclide for the appropriate geometry and dividing by the sample weight. These estimates are approximate and will vary depending upon the specific counting times and delay between sample collection and analysis.

Nuclide	Half-life	Geometry 1	Geometry 2	Geometry 3	Geometry 4	Geometry 5
Ag-110 m	253 d	380	410	240	240	120
Ba-140	12.8 d	-	-	-	-	-
Be-7	53.6 d	-	-	-	-	-
Ce-141	32.4 d	-	-	-	-	-
Ce-144	284 d	1700	2000	1000	890	660
Co-56	77 d	2600	2800	1600	1600	750
Co-57	270 d	250	300	150	140	82
Co-58	71 d	3400	3400	1900	1900	980
Co-60	5.26 y	130	130	74	78	40
Cr-51	28 d	-	-	-	-	-
Cs-134	2.07 y	150	160	93	940	45
Cs-137	30.2 y	110	110	63	61	32
Eu-152	12.5 y	700	720	440	430	200
Eu-155	1.81 y	420	480	220	230	98
Fe-59	44.6 d	57000	56000	32000	30000	15000
I-131	8.06 d	-	-	-	-	-
La-140	40.2 h	-	-	-	-	-
Mn-54	313 d	230	230	140	135	65
Nb-95	35.1 d	-	-	-	-	-
Ru-103	39.6 d	49000	53000	31000	30000	15000
Ru-106	1.0 y	1700	1800	1000	1000	470
Sb-124	60.2 d	800	880	500	490	250
Sb-125	2.7 y	300	330	180	180	91
Zn-65	244 d	730	680	370	383	190
Zr-95	64 d	7900	8200	4900	4900	2400

Appendix 4. Detection limits (Bq/kg) for each radionuclide and geometry assumming a counting time of 10^5 sec for analyses performed one year subsequent to sample collection.

POOR

QUALITY

ORIGINAL